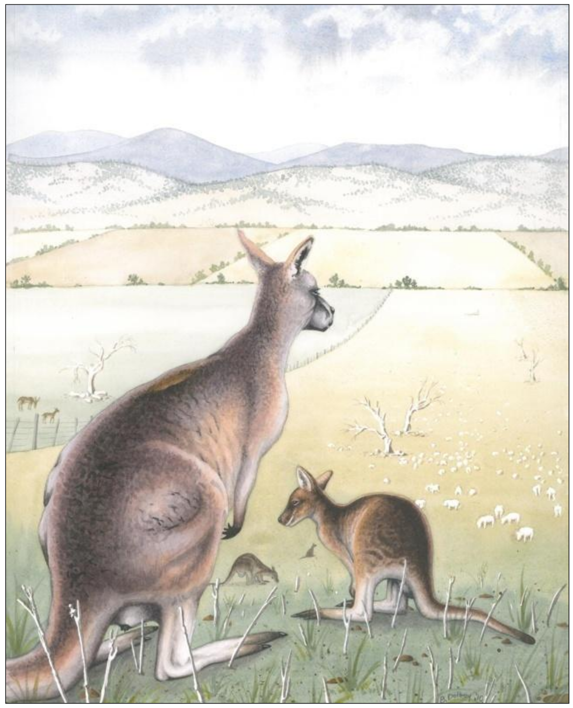
Impacts of Wildlife Grazing on Pastures in the Midlands, Tasmania



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This thesis is submitted in fulfilment of the requirement for the degree of Doctor of Philosophy at the University of Tasmania

Launceston
August 2012

UNIVERSITY OF TASMANIA

Statements and Declarations

Statement of Originality

This thesis contains no material which has been accepted for a degree or diploma by the University or any other institution, except by way of the background information and duly acknowledged in the thesis, and to the best of my knowledge and belief no material previously published or written by another person except where due acknowledgement is made in the text of the thesis, nor does the thesis contain any material that infringes copyright.

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Statement of Co-Authorship

A published refereed paper, based on some of the research presented in Chapter 3 and Chapter 4 of the thesis, is included as an appendix:

Smith RW, Statham M, Norton TW, Rawnsley RP, Statham HL, Gracie AJ, Donaghy DJ (2012) Effects of wildlife grazing on the production, ground cover and plant species composition of an established perennial pasture in the Midlands region, Tasmania. *Wildlife Research* **39**(2), 123-136.

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Statement of ethical conduct

The research associated with this thesis abides by the international and Australian codes on human and animal experimentation, the guidelines by the Australian Government's Office of Gene Technology Regulator and the rulings of the Safety, Ethics and Institutional Biosafety Committees of the University. Animal ethics approval number A0009820.

Rowan William Smith August 2012

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Abstract

Management of Tasmania's native and introduced wildlife on private land is a contentious issue for landowners, animal welfare groups and the Tasmanian State Government. In 2005 the use of the poison 1080 (sodium monofluoroacetate) to kill wildlife was banned from use on public lands and the State Government has planned to cease all use by 2015. Many farmers believe that the impact of grazing by native wildlife on pastures is significant and results in a considerable financial impost. However, only limited research has been undertaken to quantify this wildlife grazing impact. Grazing and browsing wildlife include Forester kangaroo (*Macropus giganteus tasmaniensis*), Bennett's wallaby (*Macropus rufogriseus rufogriseus*), Tasmanian pademelon (*Thylogale billardierii),* brushtail possum (*Trichosurus vulpecula*) and fallow deer (*Dama dama*).

Results of a grazing impact study in the Midlands region of Tasmania found that in the year 2009 alone, average pasture loss for the area 0-800 m from the native vegetation edge was 1,730 kg dry matter (DM)/ha. These losses of pasture decreased with increasing distance from native vegetation and varied between 0-100% depending on season and distance from native vegetation. Periodic harvests of pasture plots and collection of wildlife faecal pellets indicated shifts in grazing behaviour with reference to seasonal pasture feed availability. Pasture losses and faecal collections were lowest during spring 2009, while pasture losses were greatest during winter 2008, matching highest and lowest pasture growth rates over the experimental period.

Production of perennial and annual grasses was greater in protected plots (areas protected by grazing exclusion cages) than exposed plots (not protected by cages), while the amount of subterranean clover (*Trifolium subterraneum*) increased in 2009 in exposed plots possibly due to reduced competition from grasses. Composition of annual grasses was greater in enclosed plots in close proximity to the native vegetation and the amount of bare ground was greater in exposed plots.

Exclusion of grazing for 2 years had no significant ($P>0.05$) effect on soil health parameters such as: ammonium nitrogen, nitrate nitrogen and organic carbon levels, pH, electrical conductivity, and root biomass. Microbial analysis also indicated no significant ($P>0.05$) effect on bacterial biomass, fungal biomass, total active microbial biomass, and fungal/bacterial ratio. These results indicated that either 2 years may not have been a long enough trial period to detect changes in soil health, or that the size of exclosure treatments may have been too small to prevent buffering influence from outside the exclosure.

A study investigating the influence of grazing damage during pasture establishment found that wildlife grazing had a significant ($P<0.05$) effect on production of all 4 pasture types sown. Pasture types containing phalaris (*Phalaris aquatica*) produced the highest DM and had greater ground cover than pasture types based mainly on ryegrass (*Lolium perenne*) and cocksfoot (*Dactylis glomerata*).

Pasture biomass losses under some conditions were found to be as high as 100% within 25m and 68% within 800 m of native vegetation. However, feed availability was found to be a large determinant in the distance and direction wildlife will travel to graze. Continued exposure to wildlife grazing resulted in a higher proportion of bare ground and reduced production of annual and perennial grasses. Control of wildlife grazing during pasture establishment may be necessary to reach optimum production and protect pasture species susceptible to grazing at the seedling stage. Continued grazing of pastures by wildlife is likely to amplify the effects of drought. The results of this thesis provide important information to land owners and that can better equip them to manage wildlife not only at a property scale, but also a catchment scale.

Table of Contents

List of Tables

List of Figures

List of boxes

List of species referred to in this thesis

Wildlife nomenclature follows the Zoological Catalogue of Australia, Bureau of Flora and Fauna (Commonwealth of Australia 1988). Plant nomenclature follows 'The Student's Flora of Tasmania' Parts 1-4 (Curtis and Morris 1994). Introduced species are denoted by an asterisk.

Wildlife

- Common brushtail possum *Trichosurus vulpecula* (Kerr, 1792)
- Tasmanian pademelon *Thylogale billardierii* (Desmarest, 1822)
- Red-necked wallaby *Macropus rufogriseus* (Desmarest, 1817) - Tasmanian sub-species Bennett's wallaby *M.r. rufogriseus* (Desmarest, 1817)
- Eastern grey kangaroo *Macropus giganteus* (Shaw, 1790) - Tasmanian sub-species Forester kangaroo *M.g. tasmaniensis* (Le Souef, 1923)
- Fallow deer* *Dama dama* (Linnaeus, 1758)
- Common wombat *Vombatus ursinus* (Shaw, 1800)
- Rabbit* *Oryctolagus cuniculus* (Linnaeus, 1758)
- Tasmanian bettong *Bettongia gaimardi* (Desmarest, 1822)
- Long-nosed potoroo *Potorous tridactylus* (Kerr, 1972) - Tasmanian sub-species *Potorous tridactylus apicalis*
- Eastern barred bandicoot *Perameles gunnii gunnii* (Gray, 1838)
- Spotted-tailed quoll *Dasyurus maculatus maculatus* (Kerr, 1972)
- Wedge-tailed eagle *Aquila audax fleayi* (Condon & Amadon, 1954)
- Masked owl *Tyto novaehollandiae castanops* (Gould, 1837)
- Swan galaxia *Galaxias fontanus* (Fulton, 1978)

Tree and shrubs

- White gum *Eucalyptus viminalis*
- Black gum *Eucalyptus ovata*
- Black peppermint *Eucalyptus amygdalina*

- She-oak *Allocasuarina verticillata*
- Cabbage gum *Eucalyptus pauciflora*
- Gorse* *Ulex europaeus*
- Blackberry* *Rubus fruticosus*
- Wattle *Acacia spp.*
- Prickly box *Bursaria spinosa*

Pasture plants

- Perennial ryegrass* *Lolium perenne*
- Phalaris* *Phalaris aquatica*
- Subterranean clover* *Trifolium subterraneum*
- Cocksfoot* *Dactylis glomerata*
- Kangaroo grass *Themeda triandra*
- Wallaby grass *Austrodanthonia spp.*
- Silver tussock *Poa labillardierei*
- Speargrass *Austrostipa spp.*
- Weeping grass *Ehrharta stipoides*
- Soft brome* *Bromus hordeaceus*
- Silver grass* *Vulpia bromoides*
- Barley grass* *Hordeum murinum*
- Winter grass* *Poa annua*
- Lucerne* *Medicago sativa*
- Annual ryegrass* *Lolium multiflorum*
- Arrowleaf clover* *Trifolium vesiculosum*
- Balansa clover* *Trifolium michelianum*
- Sedge *Carex spp.*
- Rush *Juncus spp.*
- Sagg *Lomandra longifolia*
- Tall fescue* *Festuca arundinacea*
- White clover* *Trifolium repens*
- Browntop* *Agrostis capillaris*

List of acronyms

▪ TIA	Tasmanian Institute of Agriculture
▪ DPIPWE	Department of Primary Industries, Parks, Water and Environment
▪ BoM	Bureau of Meteorology
▪ TFGA	Tasmanian Farmers and Graziers Association
▪ TCFA	Tasmanian Community Forest Agreement
▪ FT	Forestry Tasmania

Chapter 1: Introduction

1.1 Introduction

The sustainable management of agricultural landscapes presents serious challenges for property owners, land managers and those concerned with environmental stewardship in many regions of the world (Millennium Ecosystem Assessment 2005). The clearing and modification of natural landscapes for agriculture typically results in many biophysical impacts including biodiversity loss, and may significantly change ecosystem processes and ecological interactions at multiple scales (Freemark 1995; Craig *et al.* 2000). Myriad examples are published in the scientific literature where ecosystem modification has led to unanticipated environmental change that may threaten the sustainability of a system and challenge current notions of environmental management (Mitchell and Craig 2000). Ecosystem perturbation for agricultural production may result in human-wildlife conflicts. For example, agricultural production losses for producers of US$11,076 per year (and increasing) result from predation of livestock by wolves (Muhly and Musiani 2009). Losses may also be due to grazing of improved pastures and browsing of plantations by vertebrate herbivores (Freemark 1995). Lacey *et al.* (1993) estimated that big game such as mule deer (*Odocoileus hemionus*) and elk (*Cervus canadensis*) may cost landowners in Montana, USA on average US$6,467 per annum due to forage consumption, damage to haystacks, and the costs associated with the erection and maintenance of fences around haystacks and pastures. In Uttarachal, India, damage to crops including potatoes, kidney beans and apples by wildlife such as wild boar (*Sus scrofa*) and monkey (*Presbytes entellus*) resulted in projected losses of US$15,389 per village, while livestock killed by leopard (*Panthera pardus*) added a further US$29,272 per village (Rao *et al.* 2002). Human-elephant conflicts are common in Africa where agriculture is expanding. Elephants (*Loxodonta africana*) can damage crops, food stores, water storages, and fences (Hoare 1999).

In Australia, there are many introduced and native species that may threaten the sustainability of agricultural landscapes by reducing agricultural production, damaging native vegetation and impacting on biodiversity (Hone 2007; Australian Government 2010). For example, feral pigs (*Sus scrofa*) may eat lambs (Choquenot *et al.* 1997), and damage crops, fences and pasture (Choquenot *et al.* 1996). Rabbits (*Oryctolagus cuniculus*) reduce pasture production, damage vegetation which can lead to soil erosion, and threaten native mammals through competition for resources (Williams *et al.* 1995). Foxes (*Vulpes vulpes*) may prey on lambs (Greentree *et al.* 2000) and native fauna (Saunders *et al.* 1995). In addition, mice (*Mus domesticus*) damage grain crops (Caughley *et al.* 1998), black rats (*Rattus rattus*) eat fruit in orchards (Caughley *et al.* 1998), dingoes (*Canis lupus dingo*) and wild dogs (*Canis lupus familiaris*) predate on livestock and native fauna, as well as harbour disease (Fleming *et al.* 2001) while feral horses (*Equus caballus*) and feral goats (*Capra hircus*) can compete for feed and water with livestock and potentially harbour exotic diseases (Dobbie *et al.* 1993; Parkes *et al.* 1996). In Tasmania, native marsupials such as Bennett's wallaby (*Macropus rufogriseus*), brushtail possum (*Trichosurus vulpecula*) and Tasmanian pademelon (*Thylogale billardierii*) may significantly reduce pasture production through grazing (Statham and Rayner 1995). In 2007/08 some $122 million was spent by the Australian Federal Government, as well as State and Territory governments, and land owners on the management, administration and research of vertebrate pests. The estimated costs from bird damage, alone, to agriculture and horticulture during this period was estimated to be $313 million (Gong *et al.* 2009).

In many cases, native wildlife and some introduced animals are formally protected by governments around the world for public good. However, some land owners may consider the protection of wildlife to be a direct cost to farmers. For example, land owners in the Cypress Hills, Canada reported few benefits of elk on their land and generally considered elk to be a public resource maintained at their cost (Hegel *et al.* 2009). In some cases, the costs associated with wildlife can be offset by royalties paid by hunters to hunt on a property. However, the royalty payments rarely exceed the estimated cost of impacts associated with the presence of wildlife species on farm land (Hegel *et al.* 2009). This situation may lead to varied perceptions regarding the 'worth' of the wildlife. Landowners meeting the costs of repairing fences and dealing with crop damage can have a vastly different view of wildlife than trophy hunters or urban dwellers hoping to view wildlife during recreational activities. This may lead to conflicting views and interests with regard to the management of wildlife in agricultural landscapes (Decker and Purdy 1988). The conflicting views about managing wildlife reported by Hegel *et al*. (2009) in North America are not dissimilar to those that have emerged in Tasmania and other regions of southern Australia during the past few decades as the population density of macropods has increased and placed further browsing pressure on native vegetation and farm pastures (e.g. ACT Government (2009).

1.2 Wildlife management in Tasmania's agricultural landscapes

The Tasmanian pademelon and Bennett's wallaby are the two most abundant macropodid species in Tasmania (Sprent and McArthur 2002). Forester kangaroo (*Macropus giganteus*) are locally abundant in the low rainfall areas of central north eastern Tasmania (Tanner and Hocking 2001). Brushtail possum and fallow deer (*Dama dama*) are also abundant in many areas of Tasmania. Possums are common in forested and urban areas, but particularly abundant in open forest and woodland (How 1983). The pademelon is considered to be most abundant in areas where grassy clearings adjoin shelter vegetation, and the species is usually found foraging within 100 m of vegetation where it can shelter (Johnson and Rose 1983). Wallaby are found in less dense eucalypt forests with nearby grasslands and where tall heath may dominate the landscape (Calaby 1983).

These wildlife species are hereafter referred to in this thesis as pademelon, wallaby, kangaroo, possum, and deer.

Approximately 1.5 million ha of land or 20% of Tasmania's land area is used to support extensive and intensive agriculture (Norton 2010). Much of the Tamar and Derwent river valleys, Midlands and east coast of Tasmania was cleared for pastoral enterprises in the early 1800's (Morgan 1992; Tanner and Hocking 2001). However, rocky hills, ridges and many other sites poorly suited for agriculture continue to support native vegetation in these landscapes (Statham and Rayner 1995). As a consequence, much of the farm land in Tasmania either borders intact native vegetation or supports patches of remnant vegetation. Moreover, the intensity of agricultural land use has increased in Tasmania over the past 2-3 decades, leading to an increase in both variegation and fragmentation of native vegetation in agricultural regions (Norton 2010).

Many wildlife species require vegetation for shelter from weather and predators (Tanner and Hocking 2001). Often wildlife must move from shelter into more open areas to feed. This is often the case for Tasmania's macropod species as they use native vegetation and forestry plantations for shelter and exploit areas of improved pastures as a food source. The increased modification of Tasmania's landscapes, on-going expansion of improved pastures and reduced predation pressure are thought to be some of the key drivers of increased populations of grazing wildlife (Driessen and Hocking 1992). Modification of native grasslands, clearing of native vegetation, and introduction of improved pastures for livestock grazing has been to the detriment of many marsupials, with apparently only some of the larger kangaroos benefiting from changes in habitat (Newsome 1971; Newsome 1975). Euro kangaroos (*Macropus robustus*) are one species that increased in numbers following sheep introduction into the Pilbara District of the north-west of Western Australia (Newsome 1971).

Wildlife surveys have been undertaken by the Tasmanian Department of Primary Industries, Parks, Water and Environment (DPIPWE) along selected roads in Tasmania since 1985 to help estimate numbers of wildlife and variation in population size over time (Driessen and Hocking 1992). The distribution of survey transects across Tasmania is presented in Figure 1.1. A linear regression analysis of the natural log of the data indicated there was a significant increase in the observed number of Tasmanian pademelon ($r^2 = 0.675$; $P<0.01$) and Bennett's wallaby ($r^2 = 0.547$; $P<0.01$) since 1984-85 (Figure 1.2), but brushtail possums ($r^2 = 0.255$; $P = 0.219$) had not significantly increased. All species significantly ($P<0.05$) increased between 1984-85 and 1994-95. This apparent increase in population has been attributed to the drought breaking in 1982/83, decreased hunting pressure and land use changes resulting in an increase in favourable habitat (Driessen and Hocking 1992). However, based on the available transect data, since 1994-95 the numbers of Bennett's wallaby ($r^2 = 0.175$; $P = 0.533$) and Tasmanian pademelon

($r^2 = 0.276$; $P = 0.319$) have not significantly changed, and brushtail possum ($r^2 = 0.745$; $P<0.005$) numbers have declined.

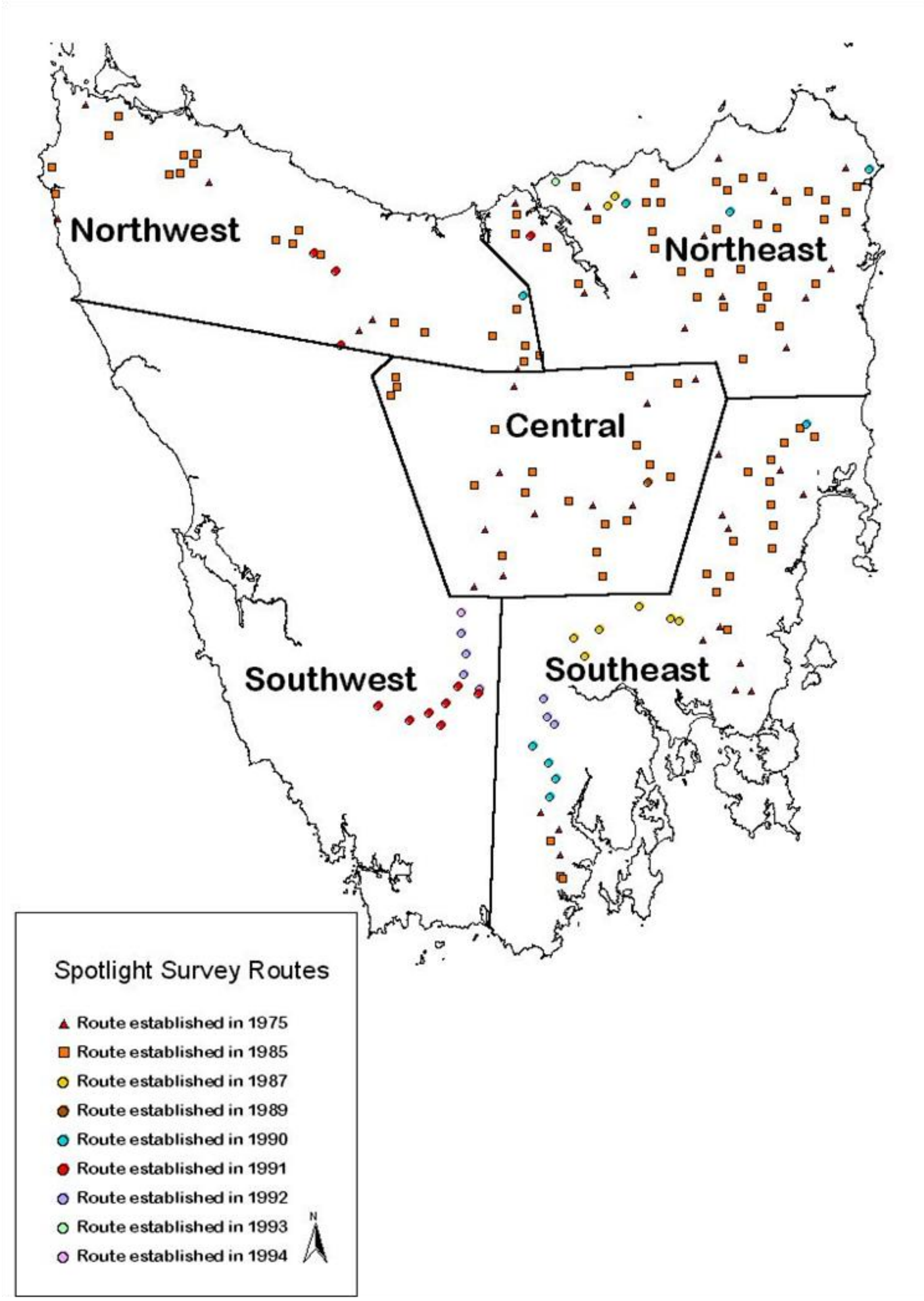


Figure 1.1 Distribution of annual spotlight survey transects conducted by DPIPWE in Tasmania**.**
Source: Greg Hocking, Principal Wildlife Management Officer, DPIPWE, 2010.

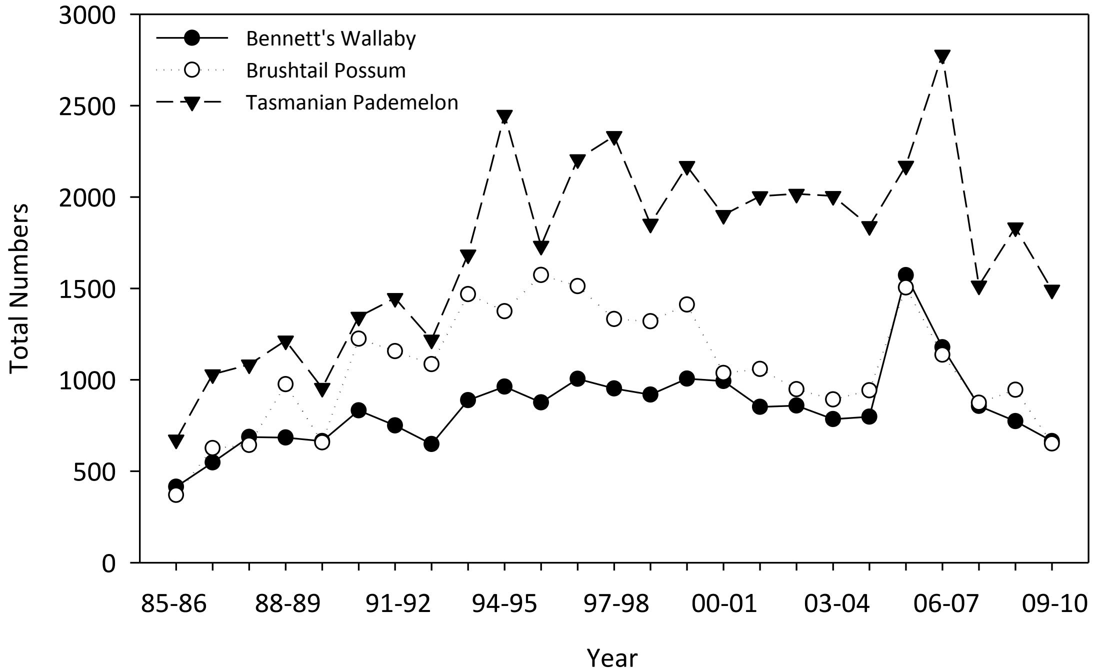


Figure 1.2 Total number of Bennett's wallaby, brushtail possum, and Tasmanian pademelon counted in 132 road-side transects conducted across Tasmania since 1985 by DPIPWE. Adapted from Statham and Statham (2010).

These survey data suggest that from the mid 1990's the overall population of Bennett's wallaby and Tasmanian pademelon has stabilised at a considerably higher level than observed during the 1980's. The total number of each species in 2006 appears to be more than double that of numbers recorded in the early 1980's (Dawson and Hocking 2006).

Pademelon, wallaby and possum have been suggested to cause the most damage to crops and reduce pasture biomass within Tasmania (Coleman *et al.* 1997a). Kangaroo and introduced deer can also cause damage to pastures, crops and fences in certain areas (Coleman *et al.* 1997b), while introduced rabbits have been shown to have a detrimental effect on sheep production (Fleming *et al.* 2002) and brown hare (*Lepus europaeus*) can browse tree seedlings (Coleman *et al.* 1997b). The common wombat (*Vombatus ursinus*) also consumes grasses (Evans *et al.* 2006; Triggs 2009), although the population density of the species is generally considered low and it may have limited effect on net pasture production (Statham and Statham 2009). Wombats digging under wallaby-proof fences can provide access points for wallabies, leading to reduced pasture biomass and increased maintenance of fences (Statham and Statham 2009). Smaller native marsupials such as Tasmanian bettongs (*Bettongia gaimardi*) and long-nosed potoroos (*Potorous tridactylus*) are thought to have little or no impact on pasture production as they mainly consume fungi (Claridge *et al.* 2007).

The impacts of grazing wildlife are mainly assessed visually by a land owner and are often most noticeable close to native vegetation, where wildlife may concentrate their feeding (Statham and Rayner 1995). Observing shorter grass close to the vegetation edge is often the only indicator that landowners use to measure wildlife grazing. Land owners may also notice longer grass growing within newly-established tree guards or around fallen tree branches close to vegetation where pademelons cannot readily access the pasture. The grazing effects of wallabies are less noticeable as they graze on the move and over greater areas (Statham and Rayner 1995). Unfortunately, few land owners record grazing damage as it is perceived as time consuming and difficult to measure.

A small number of Tasmanian studies have attempted to quantify the impacts of wildlife grazing in a defined area using exclusion cages of some form. Statham and Raynor (1995) reported losses of between 17 and 100% of pasture production based on the use of exclusion cages, and Statham (2000) demonstrated an increase in carrying capacity of sheep of over 35% on dryland pastures in the North East and South East of Tasmania when wallabies were excluded. In a short term, preliminary study of the effects of wildlife on pasture growth at Elliott in North West Tasmania, Donaghy and Tegg (2001) reported that pasture dry matter (DM) production was reduced by 21% in pastures not receiving irrigation (dryland pastures) and 34% in pastures under irrigation. Hence, it is clear that grazing by wildlife species such as pademelon and wallaby can result in significant pasture losses in some instances. Further studies were warranted to develop a more in-depth understanding of the spatial and temporal patterns of wildlife grazing.

Mitigating the impacts of wildlife grazing generally requires the use of exclusion methods such as fencing and/or lethal control methods such as the use of shooting and poison. Exclusion fencing has been used to protect pasture and crops from mainly pademelon and wallaby. It is much more difficult and expensive to exclude possum, kangaroo and deer as they can either climb or jump standard wallaby wire fences (Statham and Statham 2009). The initial financial outlay of the fence is often a barrier to land owners and the time and effort required for maintenance of the fence is often underestimated, leading to its degradation.

Lethal control of wildlife in an attempt to reduce competition with livestock for food resources may have begun with kangaroos when the pastoral industry was expanding into the eastern half of Tasmania in the 1920's (Tanner and Hocking 2001). Since its introduction for rabbit control in 1952, compound 1080 (sodium monofluoroacetate) has been the only toxin used to reduce damage caused by wallaby, pademelon and possum (Statham 2005). Poisoning with 1080 has become increasingly unacceptable to animal welfare groups and the broader community. Today, shooting is the most common method of population control with over 1 million pademelons and wallabies shot each year (Statham *et al.* 2010). Trapping has been employed mainly on plantation forestry within Tasmania to reduce populations of pademelon and possum. Trapping may be an effective method of control for wildlife considered difficult to shoot. Coleman *et al.* (1997a) produced a manual to assist landowners with managing grazing by mammalian wildlife in Tasmania. More recently, science-based options to manage wildlife in Tasmania's agricultural landscapes have been developed and evaluated as a result of the "Alternatives to the Use of 1080" program outlined below.

1.3 The "Alternatives to the use of 1080" program

Pademelon, wallaby, kangaroo, possum, and deer are seen by farmers as the major wildlife species reducing pasture biomass and causing damage to crops in Tasmania's agricultural landscapes. In these settings, improved pastures often border native vegetation that provides habitat for these wildlife species. This often provides an ideal environment for the grazing wildlife to flourish and many agricultural regions now support significant populations of these species. In many cases, the control of grazing wildlife is necessary to reduce pasture and crop losses and maintain farm productivity. Land owners use control methods such as shooting, trapping, fencing and poisoning to mitigate the effects of grazing wildlife.

In Tasmania, the use of 1080 poison for controlling wildlife is controversial. There is significant opposition from animal welfare and conservation groups to the use of 1080 in controlling browsing and grazing wildlife (Coleman *et al.* 2006). Arguments highlighting the in-humaneness of 1080 poisoning are based on the extended time until death and pain or distress an animal may experience prior to death (Sherley 2007). Death can occur in less than one hour, but may take up to several days for some species, and animals may also injure themselves during convulsions or seizures prior to death (Sherley 2007). The impact on non-target animals has also received widespread criticism (Coleman *et al.* 2006), in particular secondary poisoning in dogs (Goh *et al.* 2005).

Following increased community concern, in 2004 the Tasmanian Government announced that from December 2005 the use of 1080 would be banned on public land (Coleman *et al.* 2006). In May 2005 the Australian and Tasmanian Governments jointly announced the 'Tasmanian Community Forest Agreement' (TCFA). Within the agreement, $250 million was committed to assist in stimulating growth within the forestry industry and forestry jobs while protecting old growth forests. As detailed in clause 39 of the TCFA, $4 million was dedicated to research, field testing and demonstration of alternative control measures for controlling of browsing animals. As a consequence, a program entitled "Alternatives to the use of 1080" was established. The intention of this investment was to further reduce the amount of 1080 use in Tasmania on private lands (Coleman *et al.* 2006).

In 2007 the (then) Tasmanian Institute of Agricultural Research (TIAR) was successful in receiving funding under the "Alternatives to the use of 1080" program. The project title was '*New decision support tools to quantify and monitor the impact of herbivory of native wildlife on pasture and objectively identify alternative control mechanisms*'. The main aims of the project were to monitor and measure grazing damage on pastures and to develop tools that could predict and estimate possible damage and economic costs of grazing on properties across Tasmania. Within this project I undertook a doctoral study to address some of the key concerns surrounding wildlife grazing and its potential impacts on pastures using a case study region in the Midlands of Tasmania.

1.4 Aims of the thesis

The general aim of the thesis was to quantify and monitor the impact of native herbivores on farm pastures, and to provide scientific information for the management of grazing wildlife. The results of the study were intended to allow landowners to objectively determine the extent and degree of impact of herbivory on farm pastures, and to identify appropriate ways to control native wildlife where the quantified impacts were considered excessive. A case study was undertaken on Fosterville, a wool producing property in the Midlands region of Tasmania. The project combined models of pasture production to quantify and determine the significance of grazing on introduced pastures. This information provided the necessary basis to undertake objective benefit-cost analyses of alternative control measures for native herbivores.

The specific aims of the thesis were to:

a) Quantitatively investigate wildlife grazing on established pastures by testing for correlations between pasture loss to grazing and distance from native vegetation, and season;
b) Test if a relationship exists between observed grazing damage and an index of feeding activity by native herbivores;
c) Quantify the effects of wildlife grazing on pasture species composition and ground cover in pastures;
d) Test for correlations between wildlife grazing and soil health and root biomass in pastures;
e) Quantify the impacts of wildlife grazing on establishing pastures and measure the effects of wildlife controls on pasture production; and
f) Quantify and evaluate the economic costs of wildlife grazing in the Midlands region of Tasmania.

1.5 Structure of the thesis

Chapter 2 introduces the case study at Fosterville. Descriptions of the region and property including vegetation, land use, climate and soils are provided together with the capabilities of the property in terms of pasture growth, stock carrying capacity and animal production. The status and management of wildlife on the property and within the region are outlined. The findings of wildlife surveys conducted on the property during the experimental period to estimate the grazing distribution of wildlife species within improved pastures are summarised.

Chapter 3 details the experiment investigating the effects of wildlife grazing on established perennial pastures at Fosterville. Pasture losses in paired grazing exclusion cage plots are quantified and the relationship between distance from native vegetation and pasture loss is examined. A relationship between wildlife feeding activity and grazing damage, and some basic economic costs of wildlife grazing are described. Chapter 4 builds on the previous chapter by examining the impacts of wildlife grazing on botanical composition of the pasture sward. Changes in pasture species composition over time and the effect of wildlife grazing on ground cover are considered. Chapter 5 investigates the effect of wildlife grazing on another important component of the ecosystem - soil health. This chapter measures root biomass and soil health indicators including levels of: ammonium nitrogen, nitrate nitrogen, organic carbon, soil pH, and electrical conductivity.

Chapter 6 evaluates the effect of wildlife grazing on the establishment of pastures, and considers options for mitigating wildlife grazing impacts on pasture during the establishment period. Although relatively long in length, this chapter presents a significant body of closely inter-related results on pasture establishment that are best examined together. Chapter 7 quantitatively examines the effects of removing and introducing wildlife control through fencing on production of a newly established pasture.

Chapter 8 synthesises the major findings of the study. The chapter examines the economic implications of wildlife grazing on pasture production and the use of this information for more informed management of wildlife at a property level. There is a discussion on the benefits of this research as part of the Alternatives to the use of 1080 program. The discussion outlines opportunities for future research in the area of wildlife management on farms.

1.6 Some definitions

Browsing and grazing

Browse is defined by Sanson (1978) as ‘soft, unabrasive, low fibre herbage’ and graze ‘consists mainly of abrasive, siliceous grasses often of high fibre content’. Therefore, in pastoral systems, the damage caused by grazing wildlife may be incorrectly referred to as browsing damage. However, the diet of browsing wildlife may include plants such as small trees, shrubs, bushes, broad-leafed forbs and herbs. In contrast, grazing wildlife mainly eat grasses. The fact that both pademelon and wallaby feed on grasses and forbs in varying amounts further complicates the issue. Furthermore, according to Sanson (1978), kangaroos and wallaby fit the grazing class while pademelon fit the browser class. Hence, in this thesis I refer to the wildlife under study as ‘grazing wildlife’ since my study principally investigates their effects on sown pastures.

Distance from native vegetation and plant cover

The area consisting mainly of grassy woodland and remnant forest is referred to as native vegetation. This is the area which wildlife retreat to during the day to rest and avoid predators. Native vegetation is used as there is a clear divide between these areas and introduced exotic pastures. Therefore the distance from native vegetation refers to the distance into the introduced pasture from the native vegetation/introduced pasture interface. Other terms such as cover and shelter have been previously used by other authors. However, I refer to plant cover with reference to assessing the botanical composition of the pasture sward and including areas of bare ground. In this thesis, the cover of plant species was defined as the percentage area that each pasture species covers when projected against the soil surface. Total ground cover was estimated by summing the cover of individual plant species at a site.

Chapter 2: Introducing the Midlands of Tasmania case study

2.1 Introduction

Three types of pasture are used by graziers in Tasmania – improved, semi-improved/natural, and native pastures. Improved pastures are found on sites where native vegetation has been replaced with improved (exotic) pasture species. These sites are often fertilised and may also be irrigated. The widespread establishment of improved pastures in Australia (including Tasmania) commenced during the 1950's (Crofts 1997). This development was facilitated by the introduction of exotic species of legumes and the use of superphosphate fertiliser (Pratley 1991). Legumes improved soil fertility through nitrogen fixation, while superphosphate was used to increase the concentration of soil phosphorus and sulphur in order to enhance pasture growth (Pratley 1991). Improved grasses include cultivars selected for vigour, growth rate, drought tolerance, and pest tolerance. Perennial ryegrass (*Lolium perenne*) is probably the most common exotic grass species used in extensive grazing in Tasmania (Knox *et al.* 2006). However, the species is not well suited to the extensive sheep grazing regions of the Midlands and East Coast regions of Tasmania, where average annual rainfall may fall below 600 mm (Knox *et al.* 2006). Phalaris (*Phalaris aquatica*) is a perennial grass that is tolerant of drought, water-logging, pest and diseases (Knox *et al.* 2006). Phalaris is often used in combination with subterranean clover (*Trifolium subterraneum)* for extensive grazing in the Midlands and East Coast. Cocksfoot (*Dactylis glomerata*) is another pasture species used for extensive grazing. Cocksfoot species grows best on well drained sites, and is tolerant of low soil fertility and insect pests.

Whalley and Lodge (1987) defined natural pastures as primarily comprising native species, but possibly containing a component of exotic plant species. Many graziers have created 'semi-improved' native pastures by over-sowing the native pasture with improved grass and legume species and applying fertiliser. Natural pastures and semi-improved pastures may be difficult to manage due to their plant species richness and variety of different growth and life cycles. However, this feature can also be advantageous as the variety of species can provide feed for introduced grazing livestock at different times of the year.

Native pastures are characterised by the presence of native herbaceous species (Wheeler 1987). They have a high plant species richness and may comprise species such as kangaroo grass (*Themeda triandra***)**, wallaby grasses (*Austrodanthonia* spp.), silver tussock (*Poa labillardierei*), speargrasses (*Austrostipa* spp.), and weeping grass (*Ehrharta stipoides*) (Whalley and Bellotti 1997; Mokany *et al.* 2006). There are more than 20 species of wallaby grasses in Tasmania, several of which are particularly common in the Derwent Valley, Midlands, and East Coast regions (Mokany *et al.* 2006). Wildflowers, sedges, rushes and mosses may occur between the clumps of grasses (Mokany *et al.* 2006). Grazing of native pastures in Tasmania has resulted in the spread of exotic plant species including soft brome (*Bromus hordeaceus*), barley grass (*Hordeum murinum*), silver grass (*Vulpia bromoides*), and winter grass (*Poa annua*). These exotic species may be particularly common in degraded pastures in the Midlands region (Mokany *et al*., 2006).

2.2 Fosterville

'Fosterville' was chosen as the main site for field studies. The property is located west of Campbell Town and northwest of Ross on Ashby Road in the Northern Midlands region of Tasmania (Figure 2.1) with central grid reference (GDA) 535117E, 5353670N), and may be found on the Campbell Town, Jacobs and Ellinthorp 1:25000 TASMAP Topographic and Cadastral maps. Fosterville, along with adjoining properties 'Bloomfield' and 'Merton Vale' span some 7,780 hectares and are owned and managed by the Foster family. The family's association with this land dates back to the 1820's (Johnson *et al.* 2006). Since the properties are managed as a single unit, they are referred to collectively here as Fosterville.

Fosterville was chosen for the diversity and large numbers of wildlife known to be grazing pastures (S. Foster, pers. comm. 2007). In addition to the improved pastures, the properties support contiguous areas of native vegetation and remnant patches of native vegetation (Figure 2.2). There is an interface spanning more than 5km between the native vegetation and pastures (Johnson *et al.* 2006). Wildlife move across this interface to graze; a landscape setting typical of many properties in the Midlands and elsewhere throughout the Tasmania. The Foster family has supported and participated in a number of scientific studies on Fosterville, including biodiversity studies and the monitoring of lowland native grassland vegetation communities. The family is progressive and believe that scientific research is an important step in understanding the ecology of the property and its implications for improved management (Johnson *et al.* 2006).

Three *in situ* experiments examining the impacts of wildlife grazing on pastures were conducted on Fosterville, and a subsidiary *ex situ* experiment was undertaken at the 'Mt. Pleasant Research Laboratories' site in Launceston.

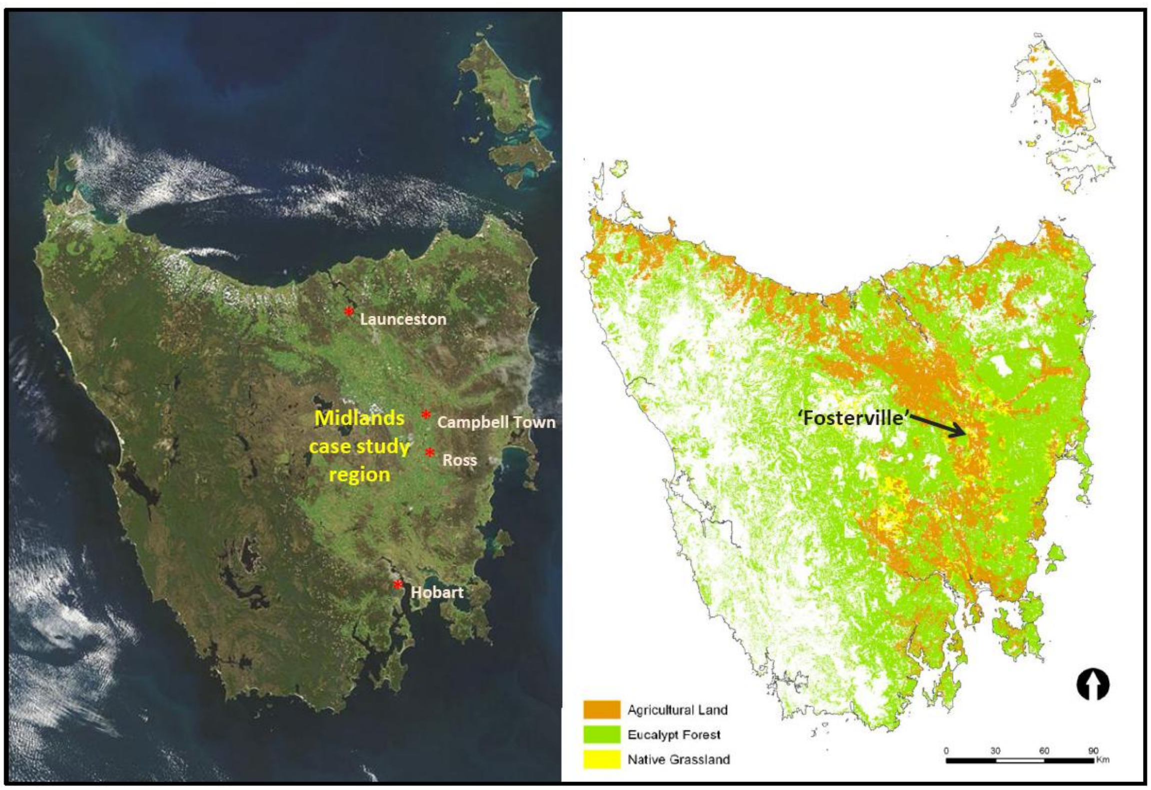


Figure 2.1 Location of 'Fosterville' (comprising the Fosterville, Bloomfield and Merton Vale group of properties) near Ross in the Midlands region of Tasmania where the case study for the field research presented in the thesis was located. Note that ex-situ experimental research was conducted at TIA facilities at Mt. Pleasant, a suburb of Launceston. Landsat ^{TM} image from 2009 (Norton 2010).

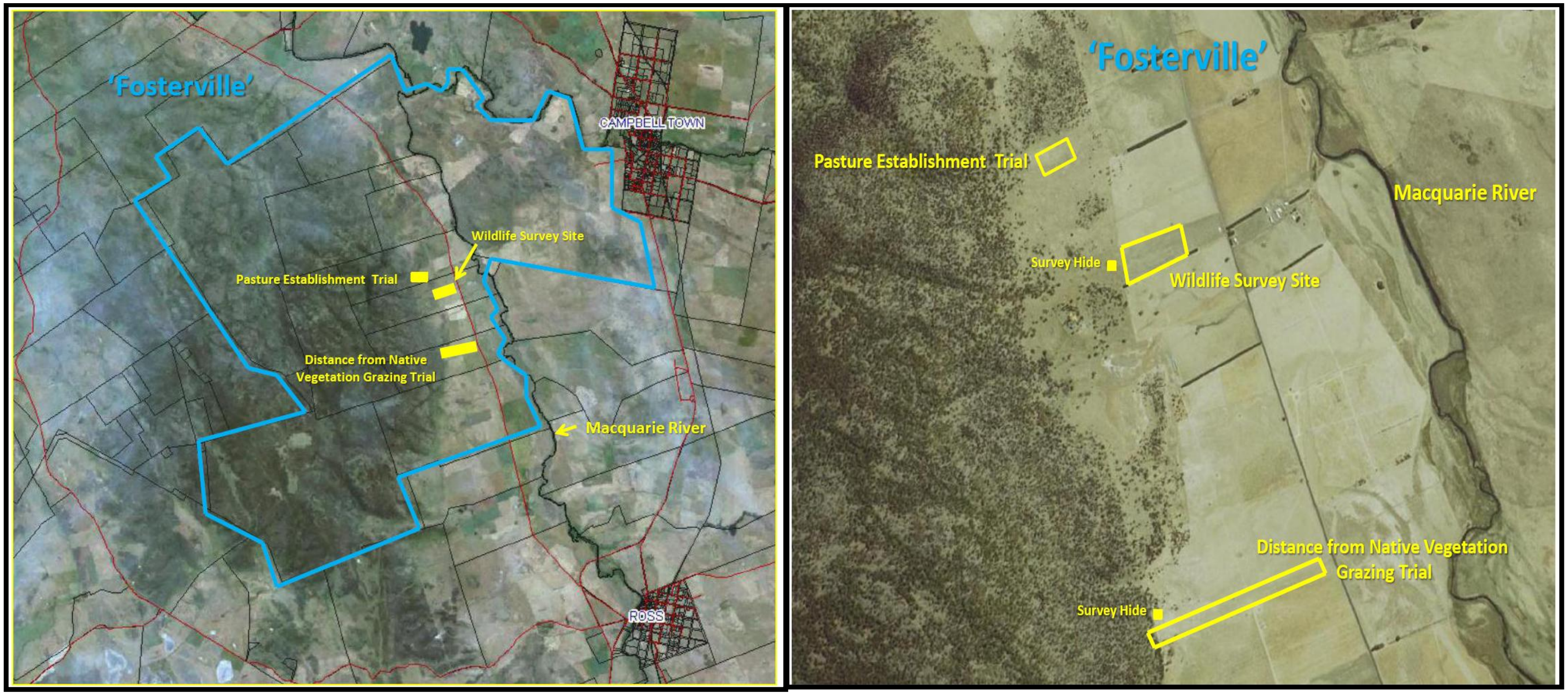


Figure 2.2 (Left) Location of 'Fosterville' near Campbell Town and Ross in the Midlands region of Tasmania (scale 1:250,000); (Right) Location of in-situ experimental trial sites and wildlife hides used on Fosterville during the two year study (scale 1: 60,000). For full details of experiments see following chapters of thesis. Satellite imagery sourced from the decision-support tool 'BITE' developed as part of the broader 'Alternatives to the Use of 1080 Program' (Norton and Lacey 2010).

2.3 Climate

The most complete and closest climate data in this region are recorded for Ross, 6 km southeast of Fosterville. January and July are the warmest and coolest months, respectively (Table 2.1). Total mean annual rainfall is 490 mm and highlights that this is one of the driest farming regions in Tasmania. Mean monthly rainfall is relatively low and ranges from 24.5 – 54.9 mm. The latest extended period of below average rainfall began in 2006 and lasted until June 2009 (Table 2.2).

Table 2.1 Climate data for Ross (The Boulevards) averaged from 1993-2010. Data obtained from the Bureau of Meteorology (BoM). Data accessed 1st October 2010.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Mean max. temperature (°C)	24.4	24.1	21.7	17.5	14.4	11.7	11.2	12.6	14.4	16.9	19.9	22.6	17.6
Mean min. temperature (°C)	10.5	10.6	7.9	5.6	3.3	1.8	1.4	2.3	3.7	4.9	6.8	8.5	5.6
Mean rainfall (mm)	47.8	39.8	32.2	37.9	24.5	38.9	40.0	48.9	54.9	42.3	45.6	38.2	490.2
Median rainfall (mm)	39.6	35.0	28.8	38.8	24.2	26.0	36.6	41.4	51.0	39.0	41.6	32.6	489.2

Table 2.2 Summary of recent (2006-2010) monthly rainfall totals for Ross (The Boulevards). Data obtained from the BoM. Data accessed 1st October 2010. *Study period ended in April 2010.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
2006	33.2	15.6	26.4	60.4	30.0	8.8	36.2	17.6	50.0	32.2	23.0	23.8	357.2
2007	44.4	13.2	32.6	5.6	62.8	12.8	30.4	35.8	30.0	25.8	17.0	32.6	343.0
2008	6.4	51.2	38.6	22.0	11.4	17.8	46.8	14.0	72.4	9.4	84.8	29.4	404.2
2009	6.6	49.4	28.8	37.2	12.2	78.8	44.6	125.6	89.2	25.8	61.4	45.0	604.6
2010*	30.2	66	56.8	43									

2.4 Soils

The Fosterville property is made up of 4 different land systems with the largest, the Isis Hills land system, covering the majority of the Macquarie Tier (Johnson *et al.* 2006). Soils on this system range from shallow (45 cm) duplex clay loam over clay soils on the higher slopes to deeper (90 cm) light clay over heavy clay soils on the flats. The majority of this land system is mapped as land capability class 5 indicating that it is suitable for grazing but not cropping. Land capability classes 6 and 7 are also found on the properties, including rocky and steep slopes that are not suitable for grazing (Johnson *et al.* 2006).

The Morningside Hills land system covers the northern most flats on Fosterville. Soil in this system ranges from shallow (45 cm) duplex sandy clay loam over medium clay on rocky crests through to deeper (100 cm) duplex sand surface over sandy clay and deep (140 cm) sandy soils on the flats (Johnson *et al.* 2006). Flats further to the south on Fosterville are made up of the Fosterville Flats and Ross Flats land systems. These deep soils (>140 cm) range from duplex clay loam to gravelly clay over medium to heavy clays, and loams to sands over clay (Johnson *et al.* 2006). There are also small isolated areas affected by salinity.

2.5 Vegetation and land use

The cleared land on the property is typical of the Midlands region. It consists of a combination of improved, semi-improved, and native pastures, and supports merino wool production as well as meat production in the form of prime lambs, mutton and beef. Small areas of irrigated fodder, particularly lucerne (*Medicago sativa*), are grown and conserved as hay or silage to feed stock over the summer/autumn/winter period when pasture growth may be insufficient for livestock requirements. Cereal crops are also grown opportunistically depending on soil moisture.

The Macquarie Tier rises 470 m above sea level on the western side of the property, 250 m higher than most of the cleared land. The Tier, that covers approximately 50% of Fosterville, is used as a bush run (see section 2.8) and is dominated by grassy white gum (*Eucalyptus viminalis*) forests and woodlands. Pockets of black gum (*Eucalyptus ovata*) and black peppermint (*Eucalyptus amygdalina*) forests occur in some areas while she-oak (*Allocasuarina verticillata*) vegetation communities are found on the drier eastern slopes (Johnson *et al.* 2006).

2.6 Improved pastures

There is a preference for growing phalaris as the main grass component on Fosterville. Phalaris is chosen as the 'core' pasture species as it tolerates drought, water-logging and pasture pests. Cocksfoot is selected for its high production, drought tolerance, low susceptibility to insect pest damage, and responsiveness to summer rains. Perennial ryegrass is only sown in small quantities as it is better suited to higher rainfall areas, is susceptible to insect pest damage and requires higher soil fertility. However, given suitable conditions, ryegrass can be productive on Fosterville. A typical combination of pasture species and sowing rate is provided in Table 2.3.

Table 2.3 Typical perennial pasture sowing rate and composition used at Fosterville.

Common name	Species	Cultivar	Sowing rate kg/ha
Phalaris	*Phalaris aquatica*	Australian	4
Cocksfoot	*Dactylis glomerata*	Currie/Porto	2
Perennial ryegrass	*Lolium perenne*	Victoca	0.5
Subterranean clover	*Trifolium subterraneum*	Goulburn	4
		Karridale	4
		Gosse	4

Fodder crops containing annual ryegrass (*Lolium multiflorum*) and arrowleaf clover (*Trifolium vesiculosum*) are also grown for fodder in the form of hay or silage. The cultivar 'Arrotas' is a late maturing cultivar of arrowleaf clover capable of growing into summer due to its deep tap-root. This can help provide herbage during the summer/autumn period when the growth of other pasture species may cease or be limited. A recent fodder crop sowing on the property is outlined in Table 2.4.

Table 2.4 Typical fodder crop sowing rates and composition used at Fosterville.

Common name	Species	Cultivar	Sow rate kg/ha
Italian ryegrass	*Lolium multiflorum*	Maverick	15
Arrowleaf clover	*Trifolium vesiculosum*	Arrotas	2
Balansa clover	*Trifolium michelianum*	Bolta	2

2.7 Semi-improved pastures

Greater production of pasture has been achieved on large areas of Fosterville by adding improved species to the native pasture (native grassland vegetation communities). Direct drilling into native pastures, rather than cultivation, has also prevented soil erosion on sandy soils. The range in botanical composition of these pastures provides feed for livestock over a longer period than otherwise would be the case (Johnson *et al.* 2006).

2.8 Native pastures

Fosterville contains some of the few remaining native pastures in the Midlands region, in particular, silver tussock and kangaroo grass grasslands that are formally recognised by the Australian Government as threatened ecological communities in Tasmania (Australian Government 2009). No fertiliser is applied to these communities and they are spelled from livestock grazing between September and March each year to allow plants to complete their reproductive cycles (Johnson *et al.* 2006). This enables pastures to persist while preventing broadleaf weeds and annual grasses from establishing. A recently compiled technical guide (Mokany *et al.* 2006) is used to inform these management decisions.

2.9 Bush runs

'Bush runs' are used for grazing livestock. They typically are located on topographically-dissected country where the terrain and native vegetation are unsuitable for clearing. These areas may comprise a range of plant species and life forms. Trees such as white gum, cabbage gum (*Eucalyptus pauciflora*) and black gum are common. Grassed woodland communities comprise silver tussock, kangaroo grass, wallaby grasses and native spear grasses. Other ground cover such as sedges (*Carex* spp.), rushes (*Juncus* spp.) and saggs (*Lomandra longifolia*) may provide shelter for stock (Mokany *et al.* 2006).

2.10 Grazing management

The livestock grazing system used on Fosterville is predominantly set stocking. Sheep may be grazed at low stocking levels in the same paddocks for long periods. For example ewes graze on improved pastures at 10-12 dry sheep equivalent (DSE)/ha for extended periods lasting up to 3 months. Rotational grazing has been utilised strategically to 'spell' some areas from livestock grazing or for other management purposes such as grazing excess pasture growth to reduce fire risk around buildings, and to manipulate pasture species composition and control weeds (Johnson *et al.* 2006).

Stocking rates vary across the property depending on land capability and pasture type. Improved pastures on the river flats support stocking rates of between 5 and 12 DSE/ha. Stocking rates on native pastures are limited to around 1.7 DSE/ha, on average, to prevent overgrazing. The major bush runs occur on the Macquarie Tier, where stocking rates are as low as 0.2 DSE/ha, on average and are spelled for up to 6 months of the year. The Foster family believe that an increase in the number of wildlife in the region is a factor restricting an increase in stocking rates in the bush runs (Johnson *et al.* 2006). In general, average stocking rates are considered to be lower than other sheep grazing properties in the Midlands region in an attempt to reduce landscape impacts.

2.11 Wildlife

The relatively intact native vegetation of the Macquarie Tier at Fosterville supports habitat for many important wildlife species that are threatened or vulnerable. These include the wedge-tailed eagle (*Aquila audax fleayi*), masked owl (*Tyto novaehollandiae castanops*), freshwater swan galaxia (*Galaxias fontanus*), eastern barred bandicoot

(*Perameles gunnii gunnii*) and the spotted-tailed quoll (*Dasyurus maculatus maculatus*) (Johnson *et al.* 2006).

Fosterville supports the habitat of Tasmania's 3 largest macropod species – the kangaroo (a sub-species of the eastern grey kangaroo), wallaby and the endemic pademelon. The population of kangaroo on Fosterville appears to have increased in recent years (S. Foster pers. comm. 2009) and may now be in excess of the 2500 estimated by Johnson *et al*. (2006). Wallaby are considered to be abundant, while the pademelon are restricted to small populations due to lack of suitable habitat. Possums are considered to be abundant and a population of deer is present in the region.

Kangaroos are specialised grazers with a diet restricted to grasses and forbs - even during times of drought when there is a shortage of suitable feed (Poole 1983). It has adapted well to grazing introduced pasture species and may compete with sheep on Fosterville (S Foster pers. comm. 2009). Kangaroos are a protected species and animal numbers are controlled on the property by shooting under a crop protection permit issued by the Game Management Branch of the Tasmanian DPIPWE.

Wallabies are common in open eucalypt forests close to open grazing areas in Tasmania. The interface between native vegetation and introduced pasture on the Fosterville property appears the favoured habitat of wallabies. Similar habitat preferences were observed by this species at Wallaby Creek in NSW (Johnson 1987). This species is partly protected and is also controlled by shooting under permit.

Pademelon are more common in higher rainfall areas of Tasmania (Johnson and Rose 1983). Small populations of the species may be found where dense vegetation (including European gorse (*Ulex europaeus*)) provides habitat in areas along the Macquarie River where it intersects Fosterville. The species is partly protected and is controlled by shooting under permit.

Possums have been implicated in rural tree decline in the Midlands region of Tasmania (Statham, 1992). A preliminary study indicated that the presence of high numbers of possums was correlated with high levels of dieback of white gum (Statham 1992). It is likely that a combination of factors including soil compaction, increased soil fertility, land clearance, drought, old age, and browsing by wildlife has led to this decline (Landsberg and Wylie 1988; Close and Davidson 2004; Kirkpatrick *et al.* 2007). Possums also favour browse over grass and a large proportion of their diet can be clover when available (Statham 1992). Gut samples analysis studies by Statham (1992) confirmed that clover can be a major component in the diet of possums. The species is partly protected and can be controlled by shooting under permit.

Deer were first introduced to Tasmania in the 1930's and the total population may now exceed 30,000 animals (Department of Primary Industries Parks Water and Environment 2010). These are found across the Central Highlands, Midlands and East Coast of Tasmania (Department of Primary Industries Parks Water and Environment 2010). Deer are primarily grazers that prefer grasses, particularly improved grasses but will also browse sedges, rushes, blackberry (*Rubus fruticosus*), wattles (*Acacia* spp.) and other native vegetation (Bentley 1983). Deer are considered to compete with sheep for pasture at Fosterville and they have been implicated in damage to crops, pasture, and tree plantings. They are also thought to contribute to the lack of regeneration of much of the native vegetation on the property (Johnson *et al.* 2006). Deer are partly protected, but may be shot during a formal hunting season. They can also be killed outside of the hunting season if they pose a threat to agricultural production. Crop protection permits may be granted to a landowner for this purpose by the DPIPWE's Game Management Services Unit (Department of Primary Industries Parks Water and Environment 2010).

2.12 Quantifying wildlife numbers and foraging at Fosterville

Developing a reliable estimate of wildlife numbers is difficult because of the varied mobility and behaviour of each species. Species such as the kangaroo and deer leave shelter vegetation each night to forage at different locations (Bentley 1983; Poole 1983), while the smaller species can graze close to thicker vegetation and may be hard to reliably observe, identify and count. When disturbed wallabies scatter individually for cover (Calaby 1983), again making them difficult to count. As a consequence, there can be a significant difference in the number of animals that a farmer or property owner thinks may be in an area compared to those that are estimated to be present using scientific census techniques, and those that are actually there (M. Statham pers. comm. 2007). Similarly, different wildlife census techniques may generate different estimates of the size of a target wildlife population due to the inherent limitations of each technique (Caughley 1977; Eberhardt 1978; Pople *et al.* 2006). Undertaking reliable and accurate measures of wildlife numbers in these landscapes are complex and expensive.

Developing a reliable population estimate for all of the wildlife species at Fosterville would be a major task that would require a significant investment of time and financial resources. Since it was both beyond the resources of the current study, and not central to its main aims, no comprehensive wildlife survey was undertaken. Rather, an attempt was made to correlate the number of animals foraging in relation to distance from the edge of native vegetation, since this information was an important context for the experimental research investigating the impacts of grazing on pastures.

Concurrent counts of wildlife were undertaken during June 2009 by observers located within hides at two Fosterville sites (pseudo-replicates) where the experimental work was to be conducted (Figure 2.2). The number of animals seen within marked zones (0-50 m, 50-100 m, 100-150 m, 150-200 m, 200-250 m, 250-300 m) at distance from native vegetation was recorded each 30 minutes during a period following half an hour after sunset until midnight or until the count of animals per zone had reached a plateau. In all, count data for sixteen observation sets were collected and analysed.

Six wildlife species were observed during the preliminary survey: wallaby, kangaroo, possum, rabbit, pademelon and deer. Quantitative (non-parametric) population models were developed for wallaby, kangaroo, possum, and rabbit using the wildlife survey count data (Box 2.1). The number of sightings of pademelon and deer were insufficient for statistical analysis. Analysis of the count data suggested that animal numbers reached a maximum within the 0-300 m region sampled, and beyond that appeared to decline in a manner consistent with the model given in Box 2.1. The location where the count for a given species reached a maximum varied with distance from native vegetation (Box 2.1, Table 2.5, and Figure 2.3).

Box 2.1 Description of population modelling of wildlife survey count data using SAS system version 9.1; source: R Corkrey, TIA, pers. comm. 2010.

A nonlinear model was fitted to accommodate the main features of the data using proc nlmixed in the SAS system (version 9.1, SAS Institute Inc., Cary, NC, USA). The analysis used the equation:

where: λ = modelled number of animals
A = controls the height of the fitted curve
B = controls where the maximum peak is along the range of distances
C = controls the rate of change away from the maximum
d = distance from native vegetation

Parameter B was significantly different between species due to the considerable variation in count recorded for each wildlife species. There was no significant difference between C parameters for each wildlife species. The C parameter was tested using a linear contrast between the parameter for each species. The observed count was assumed to have a Poisson distribution. The predicted count found a significant difference in parameters A and B for each species. The estimate of each parameter for each of these species is given in Table 2.5.

Table 2.5 Parameter estimates of each wildlife species used in the population model as well as the measure of the fit of the model, chi-square (χ^2) and P its probability.

		Parameter estimates			Overall fit			
Species	Parameter	Estimate	t Value	Pr > t	R^2 %	RMSD	χ^2	*P*
Wallaby	*A*	4.0765	24.05	<.0001	16.2	2.86	394.3	0.99
	B	13.0568	4314.16	<.0001				
Kangaroo	*A*	0.8587	16.52	<.0001	2.0	1.40	461.8	0.69
	B	202.11	466125	<.0001				
Possum	*A*	0.1590	7.79	<.0001	0.4	0.41	469.2	0.60
	B	134.03	3.275E7	<.0001				
Rabbit	*A*	0.05325	3.99	<.0001	0.7	0.24	467.5	0.63
	B	228.35	1.168E7	<.0001				
All species	*C*	0.000047	17.77	<.0001	28.7	1.61	1346.4	1.00

There were greater numbers of wallaby observed than any other species. Based on the model, the count of wallaby peaked at 4 wallabies at a distance of 13 m from the edge of native vegetation. Less wallabies were observed past 100 m. In contrast, kangaroos appeared to graze at greater distances from native vegetation. The model estimated a count peak of 0.85 kangaroos at 202 m. The count of rabbit and possum was less clear and the model suggested no relationship with distance from native vegetation. The lack of a relationship may be explained by rabbit burrows within pasture and dead eucalypt trees within pasture providing dens for possums. These results appeared consistent with the known ecology of the wildlife. For example, wallaby are considered to forage closer to native vegetation compared with kangaroo (Blumstein and Daniel 2003).

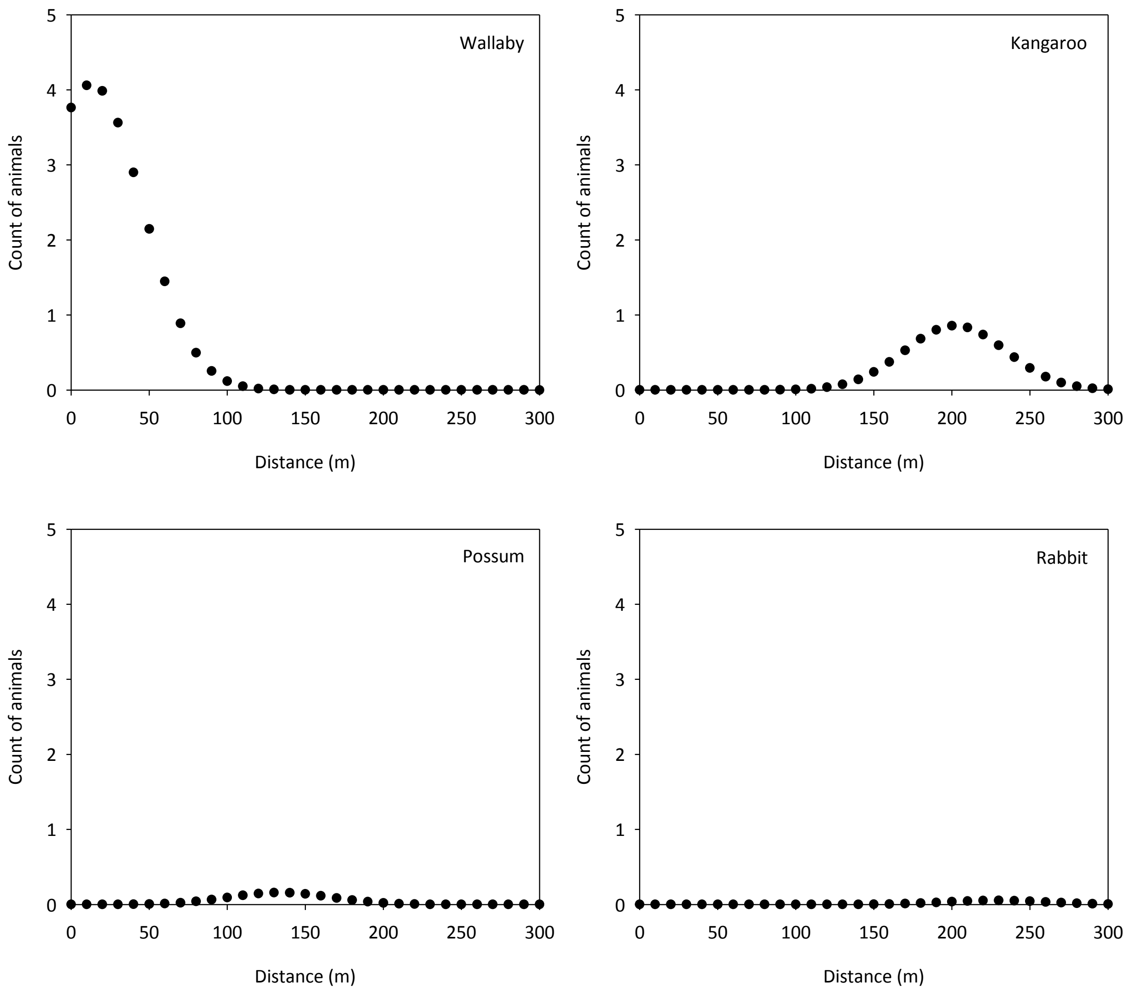


Figure 2.3 Non-linear model of count data for possum, rabbit, kangaroo and wallaby (clockwise from top left) indicates where the count peaks with distance (0-300 m) from native vegetation at Fosterville.

2.13 Challenges with the experimental design of the study

The focus of my thesis was to quantify the impacts of wildlife grazing on pastures (both established and establishing) in the Midlands region of Tasmania as a basis to improve scientific understanding of these systems, quantify the economic impost of wildlife grazing on production, and identify options to mitigate grazing impacts and improve regional wildlife management. Three *in situ* experiments examining the impacts of wildlife grazing on pastures were conducted at Fosterville, and a subsidiary *ex situ* experiment was undertaken at the 'Mt Pleasant Research Laboratories' site in

Launceston. The *in situ* experiments aimed to empirically test for relationships between wildlife grazing and several environmental variables including spatial and temporal variation in pasture production, pasture species composition, and the proximity of native vegetation that may provide a source of cover or shelter for wildlife. The *ex situ* experiments were designed to test some of the animal-plant interactions that were examined in the field, but under more controlled conditions where the species and number of individual animals involved in grazing were known and could be manipulated.

The design of the *in situ* experiments was strongly influenced (and limited) by the inherent spatio-temporal variability observed in the improved pasture systems at Fosterville, and the dynamic nature of wildlife grazing occurring on the property as a result of major differences in the relative abundance, mobility, behaviour and foraging ecology of the wildlife species under examination. In a 'perfect world', the study would have used 'true' control or reference sites to quantitatively evaluate how wildlife grazing ('the impact') had changed the site over time compared to its historical condition (e.g. (Stewart-Oaten *et al.* 1986; Wardell-Johnson and Williams 2000; Stewart-Oaten 2003). True control sites would have identical environmental conditions to the sites used to examine the impacts of wildlife grazing. The purpose of the true control sites is to enable differentiation between natural variability and stochastic factors versus those environmental trends actually resulting from the impact. Both types of sites can then be sampled and monitored with the same methods, intensity, and frequency to allow for paired comparison of their trajectories (Pickett and Parker 1994).

However, in many field-based ecological studies it may not be possible to identify true control sites and to replicate true control sites due to the inherent variability of the systems under study. This was the situation at Fosterville. As outlined earlier in this chapter, the grassland and woodland ecosystems of the Midlands have been significantly modified since European occupation (Michaels *et al.* 2010). Many native vegetation communities have been cleared and highly modified to support agriculture (Kirkpatrick and Bridle 2007). These landscape interventions have significantly changed the biodiversity and ecosystem dynamics and ecosystem function across all scales within the region (Kirkpatrick *et al.* 2007). Hence, at Fosterville it was not possible to identify true control sites or to replicate true control sites to study the impacts of wildlife grazing. Instead, pseudo-replication of control sites (e.g. sites where grazing was excluded) and impact sites (e.g. where grazing occurred) was used to examine changes in environmental conditions in pasture systems that could be attributed to wildlife grazing.

Pseudo-replication was defined by Hurlbert (1984) as the use of inferential statistics to test for treatment effects with data from experiments where either the treatments are not replicated (though samples may be) or replicates are not statistically independent. The potential limitations of pseudo-replication and related sampling methods in ecological field experiments have been relatively widely covered in the science literature and these contributions indicate the potential for erroneous interpretations and inferences that may arise using inadequate data (Eberhardt 1976; Cook and Campbell D.T. 1979; Connell 1983; Stewart-Oaten *et al.* 1986; Stewart-Oaten 2003).

Recognising the limitations associated with pseudo-replication, Conquest (2000) suggested that the results of field studies based on this approach should be interpreted with caution, professional expertise, and a thorough knowledge of the project, site, and related system processes. Conquest (2000) believed that the ability to discern environmental change due to a grazing impact, rather than data variability could be addressed through the use of a scientific framework of repeated analyses, long term datasets for spatial and temporal trends, and monitoring within a statistical framework (Pickett and Parker 1994; Parker and Pickett 1997; Wissmar and Bisson 2003; Zedler 2005). In some cases, it may also be possible to determine the sample size required to detect a certain amount of change using power analysis (Zar 1999).

My study employed a scientific framework to establish a time series of data on spatial and temporal variation in wildlife grazing at different locations on Fosterville. Every effort was made to maximise the replication of grazing ‘treatments’ and sample sites, and the length of field monitoring and sampling within the time, logistical and financial constraints of the study. The results of the field studies reported later in the thesis have been interpreted with caution and, where possible, further tested and evaluated using *ex situ* experimentation at Mt Pleasant.

2.14 Discussion

Graziers in the Midlands region of Tasmania have repeatedly called on governments to provide support to help mitigate the perceived impacts of high numbers of grazing wildlife on farm production and profitability (S. Foster pers. comm. 2009). The survey conducted of wildlife on Fosterville in this study outlined earlier in this chapter found significant numbers of several grazing wildlife species and appeared consistent with the view that the wildlife grazing pressure on pastures in this area could be substantial.

Fosterville has substantial contiguous areas of native vegetation and remnant patches of native vegetation that, together, represent a large interface between the native vegetation and pastures. The grazing wildlife move across this vegetation interface to graze and their movements may be influenced by factors such as food availability and predation (S. Foster pers. comm. 2009). My survey data suggest that the number of wildlife species and relative abundance of individual species may change across the property over time and that factors such as food quality and distance from native vegetation that provides shelter may be important drivers. This dynamic is the focus of the research presented in Chapter 3 where correlations between pasture loss to grazing and distance from native vegetation, season, and quantify how grazing pasture loss varies both spatially and temporally in perennial pastures were examined.

Overall, the climate, soils, vegetation and wildlife reported for Fosterville are indicative of a number of properties across the Midlands of Tasmania. The native grassland and woodland ecosystems of the region have been cleared and significantly modified for agriculture and, as a result, the biodiversity complement and ecosystem processes and functions naturally operating have been disrupted, modified and depleted (Kirkpatrick *et al*. 2007). As a consequence, it was not possible to identify and replicate true control sites to study the possible impacts of wildlife grazing. Instead, my in-situ studies and experiments were based on pseudo-replication of control sites and impact sites to examine changes in environmental conditions in pasture systems that could be attributed to wildlife grazing. To address the known limitations with pseudo-replication, a scientific framework was employed to establish a time series of data on variation in wildlife grazing at Fosterville and every effort was made to maximise the replication of the field research and the length of monitoring and sampling within the constraints of the study. The results of the field studies were interpreted with caution and, where possible, further tested and evaluated using *ex situ* experimentation at Mt Pleasant.

The variegated and fragmented nature of the native vegetation and pastures found at Fosterville has important implications for the interpretation of the data collected during the study, as well as the study's experimental design – especially in relation to understanding and modelling the functional response of the grazing wildlife (Caughley and Lawnton 1981; Caughley 1982). There exists a relatively large body of theoretical and empirically-based scientific literature that considers the functional response of herbivores and the implications for vegetation succession and the evolution of plant-animals interactions (e.g. (Holling 1965; Murdoch 1973; Nagy *et al.* 1990; Roughgarden 1997). The interaction or relationship between food availability and the rate of food intake by herbivores is a functional response that has been used to (qualitatively and quantitatively) model the foraging behaviour of vertebrate herbivores and vegetation ecosystem dynamics (Short 1985; Short 1986; Caughley *et al.* 1987; Fletcher 2006). However, as is discussed later in the thesis, developing a reliable understanding of the functional response of grazing wildlife species in the Midlands of Tasmania is complex. Different native, grazing wildlife species and individual animals can move relatively quickly each day between a mix of both native vegetation communities and a mix of introduced pasture-based systems to potentially meet their nutritional requirements (Short 1986; Jarman and Phillips 1989; Fletcher 2006). The diversity of food options, and access to food of variable palatability and nutritional value will almost certainly influence foraging behaviour (Short 1987; Jarman and Phillips 1989), but it was not the focus of my research, nor generally feasible to determine grazing impact across all vegetation communities and systems traversed by the wildlife under study. As a consequence, my scientific inferences from the available field data focus on understanding the implications of wildlife grazing for pasture-based systems, as opposed to both pasture-based systems and native vegetation communities and ecosystems.

A property management plan has been developed for the Fosterville properties owned and managed by the Foster family, and the findings of the research reported in the subsequent chapters of the thesis is intended to be used to further improve this management plan. It is hoped that the research findings from this thesis will be of relevance to many situations across the Midlands region and elsewhere in Tasmania where improved pastures or crops border native vegetation that shelters significant numbers of grazing wildlife.

Chapter 3: Effects of wildlife grazing on the production of an established perennial pasture

3.1 Introduction

Grazing and browsing wildlife species in Tasmania have long been implicated in causing significant damage to crops, pasture and tree plantations (Statham and Rayner 1995; Wardlaw and Burton 2008). Wallaby, pademelon and possum are believed to be the species causing the most damage (Coleman *et al.* 1997a), however species such as kangaroo and the introduced deer are also responsible for damage in some areas (Coleman *et al.* 1997a). In 1990, the Tasmanian Farmers and Graziers Association (TFGA) estimated that the cost to crops and pastures from browsing and grazing damage was worth between AU$4.5 and $6.0 million per annum (Coleman *et al.* 1997a). In addition, Forestry Tasmania (FT) estimated damage to tree plantations of AU$2.8 million per annum (Coleman *et al.* 1997a). In 2006-07, FT spent more than AU$1.0 million on browsing management in eucalypt plantations across Tasmania (Wardlaw and Burton 2008).

Population surveys conducted by DPIPWE have shown that increases in the numbers of wallaby and pademelon have occurred since the early 1980's (Figure 1.2). Land clearing, improvement of pastures, development of forestry plantations, and decrease in hunting are factors that may have led to an increase in wildlife numbers (Driessen and Hocking 1992). It is likely that previous estimates of economic losses are now outdated and are underestimates of the actual losses.

Following increased community concern about animal welfare, in 2004 the Tasmanian Government announced that from December 2005 the use of 1080 would be banned on public land (Coleman *et al.* 2006). The 'Tasmanian Community Forest Agreement' (TCFA) was jointly announced by the Tasmanian and Federal Governments in May 2005 (Coleman *et al*. 2006). Within the agreement, AU$4 million was dedicated to research, field testing and demonstration of alternative measures for control of grazing and browsing animals. The intention was to phase out the use of 1080 poison as a wildlife control measure in Tasmania by 2015 (Community Leaders Group 2001).

While some studies have assessed the influence of wildlife on pastures in Tasmania (Statham and Rayner 1995; Statham 2000; Donaghy and Tegg 2001), a need for further research was identified. A review of research into alternatives to the use of 1080 for management of browsing damage by mammals in Tasmania identified a lack of scientific data of the impacts on grazing systems (Coleman *et al* 2006). The absence of adequate data restricts the development of science-based management techniques (Coleman *et al.* 2006). The current study attempted to redress this situation by quantifying the direct influence of wildlife grazing on improved pasture production in the Midlands region.

The aims of this chapter were to:

- Quantitatively investigate wildlife grazing on established pastures by testing for correlations between pasture loss to grazing and distance from native vegetation, and season;
- Quantify how grazing pasture loss varies both spatially and temporally in perennial pastures;
- Quantify the effectiveness of the Sustainable Grazing Systems (SGS) pasture growth model to predict pasture growth on the Fosterville property, and examine the residual effects of wildlife grazing;
- Test if a relationship exists between observed grazing damage and an index of feeding activity of native herbivores; and
- Quantify and evaluate the economic costs of wildlife grazing in the Midlands region of Tasmania.

3.2 Methods

3.2.1 Experimental site

The study was undertaken at Fosterville. Details of the property are contained in chapter 2. Fosterville was chosen for the diversity and large numbers of wildlife known to be grazing pastures (S Foster, pers. comm. 2007). The property has more than 5 km of vegetation edge where improved pastures border eucalyptus forest and grassy woodlands on the Macquarie Tier (Johnson *et al.* 2006). In February 2008, a 7-strand plain wire fence was erected to give an experimental site with the dimensions of 900 m long x 50 m wide. The site ran perpendicular to, and adjoined, the Macquarie Tier forested area (Figure 3.1).

Figure 3.1 The experimental site in the foreground and the Macquarie Tier remnant vegetation in the background. Exclusion cages were placed in boundary rows of four running parallel with the native vegetation at varying distances from the native vegetation. Exclusion cages in the foreground are 800 m from the remnant vegetation edge.

The existing pasture was first surveyed in April 2008 for botanical composition of grasses, legumes and broadleaf species. Botanical composition of the sward varied considerably from broadleaf weeds and annual grasses on the upper slope to improved grasses in the mid and lower slopes, with subterranean clover the main legume throughout (see Chapter 4 for more detail). The soil type varied from a black vertosol on the flats (500-800 m from vegetation edge) to a brown dermosol (Isbell 2002) adjacent (0-300 m) to the native vegetation. The site has a slope of approximately 3%. A detailed description of the upper soil profile is provided in Table 3.1 and Table 3.2. Sheep were excluded from the site, but the fence allowed wildlife to enter and graze without impediment.

Table 3.1 Soil description of experimental site for the area 0-300 m from the vegetation edge

Horizon	Description
A1 0-140mm	dark reddish brown (5YR3/2) sandy clay loam abundant roots
B1 140-300mm	dark brown (7.5YR3/2) sandy clay abundant fine ironstone gravel
B2 >300mm	dark reddish brown (5YR3/3) heavy clay mottles, fine charcoal and pebbles, few roots
Soil fertility	Olsen phosphorus (P) 15.7mg/kg, Colwell potassium (K) 561mg/kg, mono-calcium phosphate extracted sulphur (S) 10.6mg/kg, pH (H_20) 5.4 and electrical conductivity .224dS/m.

Note: Soil description based on definitions and terminology of land surface (McDonald *et al.* 2009) and soil profile (McDonald and Isbell 2009), and soil colour characterisation keys of Munsell soil colour charts (Munsell 1973). Soil excavation ceased at 350 mm.

3.2.2 Experimental design

Paired grazing exclusion cages were placed at nine marked distance locations from the native vegetation and pasture interface. Four pairs of enclosed (caged) and exposed (uncaged) plots were located at each of the nine distances from the native vegetation/pasture interface (Appendix 1). Each plot was 0.5 m x 0.5 m in size and paired plots were located within 2 m of each other. Exclusion of grazing wildlife was achieved using exclusion cages (0.55 m x 0.55 m x 0.55 m) made from 50 mm x 50 mm mesh, which was randomly allocated to one of the paired plots and firmly fixed with tent pegs. The corners of the plots were marked with steel pegs so that the same enclosed and exposed plots areas were sampled at each assessment date. The nine distance boundaries from the pasture/native bush edge were: 25 m, 50 m, 100 m, 150 m, 250 m, 350 m, 500 m, 650 m, and 800 m. The distance between paired plots at each distance boundary was approximately 5 to 10 m. Cages were positioned in February 2008 and remained in position until the final assessment in April 2010. Pasture biomass samples were collected at regular intervals (Table 3.3).

Table 3.2 Soil description of experimental site for the area 500-800 m from the vegetation edge

Horizon	Description
A0 0-50mm	dark brown (7.5YR3/2) loam abundant roots
A1 50-170mm	dark brown (7.5YR3/2) medium clay blocky structure, abundant roots
B1 >170mm	very dark brown to black (10YR2/2) heavy clay abundant roots
Soil fertility	Olsen phosphorus (P) 17.2mg/kg, Colwell potassium (K) 144mg/kg, mono-calcium phosphate extracted sulphur (S) 10.2mg/kg, pH (H_20) 5.7 and electrical conductivity .123dS/m.

Note: Soil description based on definitions and terminology of land surface (McDonald *et al.* 2009) and soil profile (McDonald and Isbell 2009), and soil colour characterisation keys of Munsell soil colour charts (Munsell 1973). Soil excavation ceased at 250 mm.

3.2.3 Pasture measurements

Biomass accumulation

Pasture growth and biomass accumulation were measured using the paired quadrat technique described by t' Mannetje (1978). Paired quadrats (0.5 m x 0.5 m) of pasture were periodically harvested to a height of 20 mm to mimic grazing when the majority of pasture in caged areas had reached 100 to 200 mm in height. This returned the pasture to a grazed state, similar to a regular production scenario. Plots were cut with manual or mechanical hand shears and a consistent cutting height was achieved by using the top of the quadrat as a height guide. Samples were oven dried at 60 °C for 48 hours (Sanford *et al.* 2003) and weighed to give a DM yield. Following harvest, cages were placed over both paired areas and sheep were allowed to graze the experimental site for short periods (Table 3.3). Following grazing, cages were removed from the same paired area which had been grazed by wildlife previously. This 'reset' the experimental site for the next growth period. Plots were harvested a total of 10 times. Harvest dates are given in Table 3.3. This method is in contrast to Fletcher (2006) who compared the difference between pasture under an exclusion cage with a randomly chosen grazed spot a few meters from the cage. Once measured the exclusion cage was moved to the grazed spot and this was repeated bimonthly.

Table 3.3 Summary of experimental procedures carried out on the experimental site and treatment plots between March 2008 and April 2010.

Procedure	Date
Biomass assessment	2008 – 2nd June, 29th September, 19th November
	2009 – 7th January, 13th May, 11th August, 26th October, 9th December
	2010 – 15th February, 13th April
Grazing	2008 – 16th – 23rd June, 8th-16th October, 20th – 26th November,
	2009 – 15th – 20th May, 26th November – 8th December
	2010 – 18th – 22nd February

The method used by Fletcher (2006) can only measure the loss of pasture for a period of two months only. It does not take into account possible benefits of continued protection from wildlife grazing that might accrue over time. In contrast to the study of Fletcher (2006), my research used destructive methods to ensure that the staged mimicking of sheep grazing continued to affect the pasture in the absence of wildlife and that the cumulative effects of grazing over time could be examined. Destructive sampling appeared more appropriate than the use of a rising plate meter, used by Fletcher (2006) as at the beginning of the study there was a considerable amount of dry stem material. Plate meters are less accurate in the presence of stem material due to fluctuations in the resistance of the pasture (t' Mannetje 2000a). In addition, in my study, harvested samples could be hand separated into individual species to give a more accurate measure of botanical composition by dry weight.

3.2.4 Modelling of pasture growth rates using SGS pasture model

The Sustainable Grazing Systems (SGS) pasture model (Version 4.8.6), a mechanistic biophysical model developed for the Australian sheep and beef industry (Johnson *et al.* 2003), was used to simulate the growth of a dryland (non-irrigated) perennial ryegrass and subterranean clover pasture sward. The model uses daily climate data (minimum and maximum temperature, solar radiation, potential evaporation, and rainfall) to model pasture growth. The model is sufficiently versatile to simulate the range of environments represented by the pastoral regions of Australia (Cullen *et al.* 2008; Rawnsley *et al.* 2009).

Daily climate data for the experimental site were obtained from the SILO database (http://www.longpaddock.qld.gov.au/silo/, (Jeffery *et al.* 2001), which is maintained by the Queensland Environmental Protection Agency and is based on data from the Australian Bureau of Meteorology (BoM). Basic climate details for the experimental site and the experimental period are provided in Table 3.4. Pasture production was modelled on the basis of the following two management specifications: (1) pasture was defoliated daily for the 12 month period of February 2007 to February 2008 to mimic a set stocking regime on the experimental site before commencement of the experimental study and to create a steady system status within the model; and (2) the model was parameterised to mimic the experimental study, with a cutting management regime that harvested accumulated pasture herbage on the dates of biomass measurements given in Table 3.3. At each defoliation event, the pasture was defoliated to a residual of 500 kg DM/ha. The soil physical properties used in the model are given in Table 3.5. Pasture growth data modelled over the current experimental period (February 2008 to April 2010) were then compared with those measured under the exclusion cages and paired exposed plots.

Table 3.4 Summary of climate data between January 2008 and April 2010 used for pasture growth modelling in the Sustainable Grazing Systems (SGS) pasture model. Data obtained from the Bureau of Meteorology for the Campbell Town weather station.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2008 Temp max °C	27.0	22.9	23.7	17.8	15.1	13.0	10.9	12.6	14.2	18.4	19.7	20.9
2008 Temp min °C	11.3	9.8	9.0	5.8	3.6	3.2	1.0	0.9	3.3	5.0	6.8	7.5
2008 Rainfall mm	5.0	49.8	24.4	26.0	13.6	29.2	34.7	14.6	63.8	7.0	70.5	33.5
2009 Temp max °C	26.5	24.0	22.1	17.8	15.4	12.2	11.6	13.1	14.4	16.7	21.8	23.5
2009 Temp min °C	10.0	10.6	10.5	6.2	3.4	3.5	1.8	5.2	5.0	4.8	9.1	9.4
2009 Rainfall mm	4.0	35.2	39.7	35.0	16.8	57.8	53.4	134.5	95.0	21.3	58.5	43.0
2010 Temp max °C	26.1	25.5	24.1	19.1								
2010 Temp min °C	10.4	11.9	9.6	7.1								
2010 Rainfall mm	29.0	59.0	49.4	57.9								

Table 3.5 Summary of soil properties used in the pasture growth model.

Infiltration model: Capacitance				
Soil profile	Surface	A	B1	B2
Soil depth	2mm	50mm	100mm	200mm
Generic soil type	Sand	Sand	Clay	Clay
Soil physical properties				
Field capacity (-100 cm)(%v)	17	17	46	46
Bulk density (g/cm^3)	1.30	1.30	1.30	1.30
Wilting point (-150 m)(%v)	7	7	28	28
Ksat (mm/h)	54.2	54.2	1.5	1.5

3.2.5 Wildlife measurements

All wildlife grazing within the trial site were free ranging. Animal ethics was approved by the University of Tasmania's Animal Ethics Committee (Approval number A0009820). Various 'pilot' methods were used to confirm the presence and absence of wildlife species. Initially, spotlight surveys were undertaken to identify the wildlife species grazing on the experimental site. These were conducted by spotlighting from a vehicle, and from a fixed location one hour after sunset. Infrared digital scouting cameras (ScoutGuard SG550/SG530; HCO Outdoor Products, Norcross, GA) were also used within the experimental site to detect 'light shy' species such as deer. These pilot surveys showed that numbers of wildlife within the experimental site varied considerably night to night depending on weather and human activity. It was decided that due to limited resources an index of feeding activity through the collection of faecal pellets was used as alternative to spotlight surveys.

An index of feeding activity or animal presence was developed by undertaking a faecal pellet survey. Faecal pellet count plots (5 m x 5 m) were established at each distance boundary (Figure 3.2). This plot size was used as it fitted within the experimental site and was in line with the exclusion cages at each distance location allowing comparison with pasture measurements. Unfortunately, these plots could not be replicated at each distance location due to space limitations. Space also prevented using other faecal pellet count techniques such as the ones implemented by Perry and Braysher (1986), Banks (2001), and Forsyth *et al*. (2007) who used long transect lines with multiple small circular plots spaced at regular intervals to estimate abundance of kangaroos and deer respectively. These plots were first cleared of existing faecal pellets with a light leaf rake, avoiding excessive disturbance of the existing pasture and exposed soil. Each plot was surveyed on 7 occasions between February and September 2009. Individual pellets were counted rather than groups due to the difficulty in defining pellet groups from macropods as recommended by Perry and Braysher (1986). Pellets collected were identified to species (as far as possible) and dried at 60 °C for 48 hours and dry weights recorded. For the purposes of this investigation pellets of all species were summed. An average faecal pellet output (g/ha/day) was calculated using the dry weight and the number of days between collection periods. On only one occasion were the faecal collection and pasture harvest time periods identical in length. Therefore, for the purposes of the correlation, only one pasture harvest/faecal collection point was used.

Figure 3.2 Bennett's wallaby faecal pellet (left) and collection of faecal pellets within the 5 x 5 m marked collection area (right).

3.2.6 Data analysis

A linear mixed model (PROC MIXED) was used to analyse pasture biomass with distance from the bush edge. The model was fitted to the data with distance and treatment (enclosed or exposed) and distance by treatment terms treated as fixed effects. Distance by paired plots was the random term used to test the distance effect, while the other fixed effects were tested with the residual error.

Significant differences between accumulated harvested biomass (t DM/ha) of enclosed and exposed plots were determined by a paired t-test at each distance using the SPSS statistics package (IBM SPSS Inc. 2010) in Section 3.3.1.

The impact of browsing wildlife grazing on relative pasture loss with distance from the native vegetation was estimated by fitting a logistic function to the percentage pasture loss with distance from the native vegetation in Section 3.3.1. This was done using the Marquard's iterative non-linear procedure PROC NLIN of SAS version 9.1 (SAS Institute Inc. 2004).

The logistic function was defined as:

$$\frac{a}{1} + EXP\ (-b - c * DIST)$$

where L is the % pasture loss at distance, DIST, from the bush edge/pasture edge. a, b and c are parameters for the equation. The point at 50% pasture loss was determined by using the predicted equation for each sample weight.

The SGS Pasture Model (version 4.8.6) was used to simulate pasture growth rates for the Fosterville property (Johnson *et al.* 2003) in Section 3.3.2. Measured biomass data of enclosed and exposed plots were inputs along with climate data summarised in 3.2.4.

A paired samples t-test was performed in SPSS statistics package (IBM SPSS Inc. 2010) to compare total pasture biomass between enclosed and exposed plots in Section 3.3.3.

A bivariate Pearson's correlation procedure was performed using the SPSS statistics package (IBM SPSS Inc. 2010) to determine the correlation between total pasture loss, proportion pasture loss and faecal output in section 3.3.4.

3.3 Results

3.3.1 Pasture production and loss

Differences in total pasture biomass between the enclosed and exposed plots decreased with distance from edge of native vegetation (Figure 3.3). Analysis of the data using a paired t-test showed that there was a significant difference between enclosed and exposed plots at 25, 50, 100, 150, 250, 350 and 500 m from the native vegetation edge. For example, at 25 m enclosed plots yielded 12.58 t DM/ha compared with 3.95 t DM/ha in exposed plots. In contrast, at 800 m enclosed plots yielded 6.14 t DM/ha, compared with 5.39 t DM/ha in exposed plots. An example of the growth differences between enclosed and exposed treatments is displayed in Figure 3.4. It shows the greater biomass accumulation in the enclosed treatment.

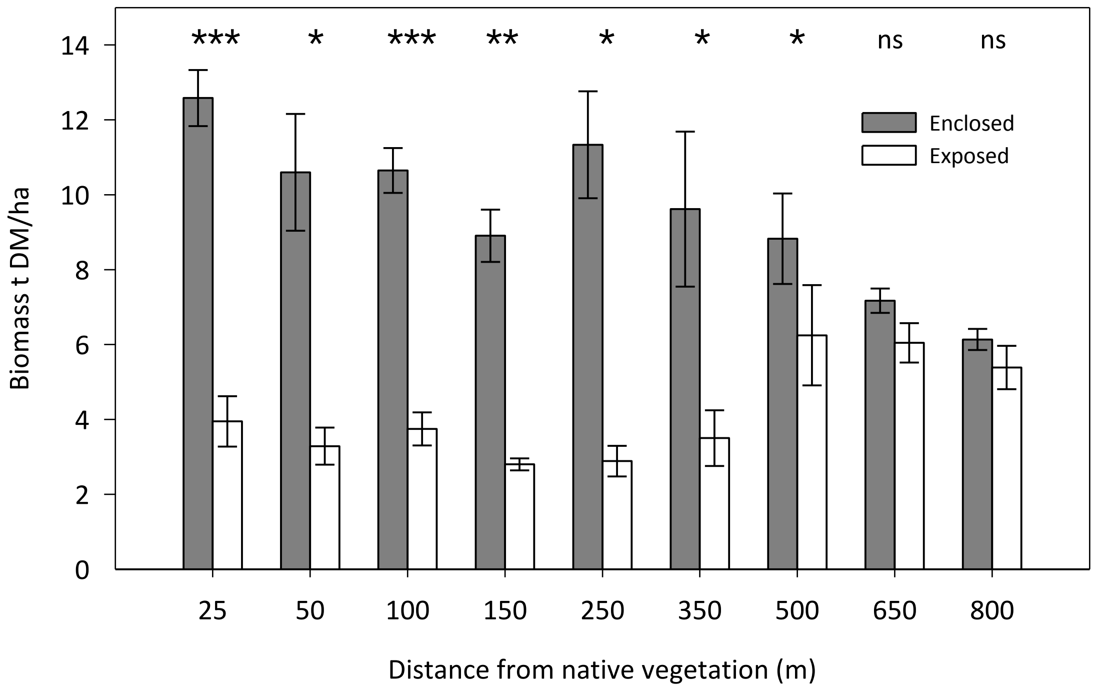


Figure 3.3 Accumulated harvested biomass (t DM/ha) of enclosed and exposed plots between February 2008 and April 2010. Error bars represent the standard error of the mean of four replicates. * represents significant (P<0.05) differences, ** significant to (P<0.01), ***significant to (P<0.001) and as determined by a paired t-test, between enclosed and exposed plots at each distance.

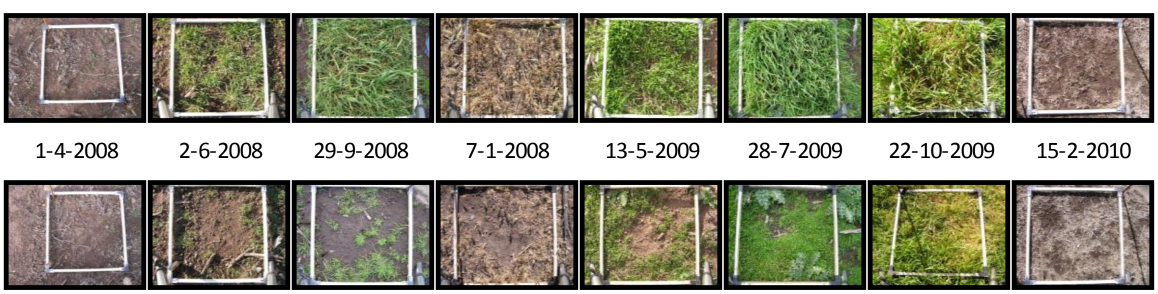


Figure 3.4 Example of differences in growth between enclosed (top) and exposed to wildlife grazing (bottom) treatments at 25 m from the native vegetation edge over the trial period.

Grazing impact was measured as a percentage reduction in the amount of biomass harvested and is summarised for 2008, 2009 and 2010 in Figure 3.5, Figure 3.6 and Figure 3.7, respectively. The reduction in biomass due to grazing decreased with increasing distance from the native vegetation edge. The relationship was best explained by a logistic regression. Changes to the parameters of the logistic regression were influenced by season. In 2008, pasture loss was least during late spring, decreasing from close to 100% at 25 m to 10% at 800 m on the fitted curve. In contrast, pasture loss earlier in 2008 remained high at 800 m, being 65% during early winter and 33% during early spring (Figure 3.5).

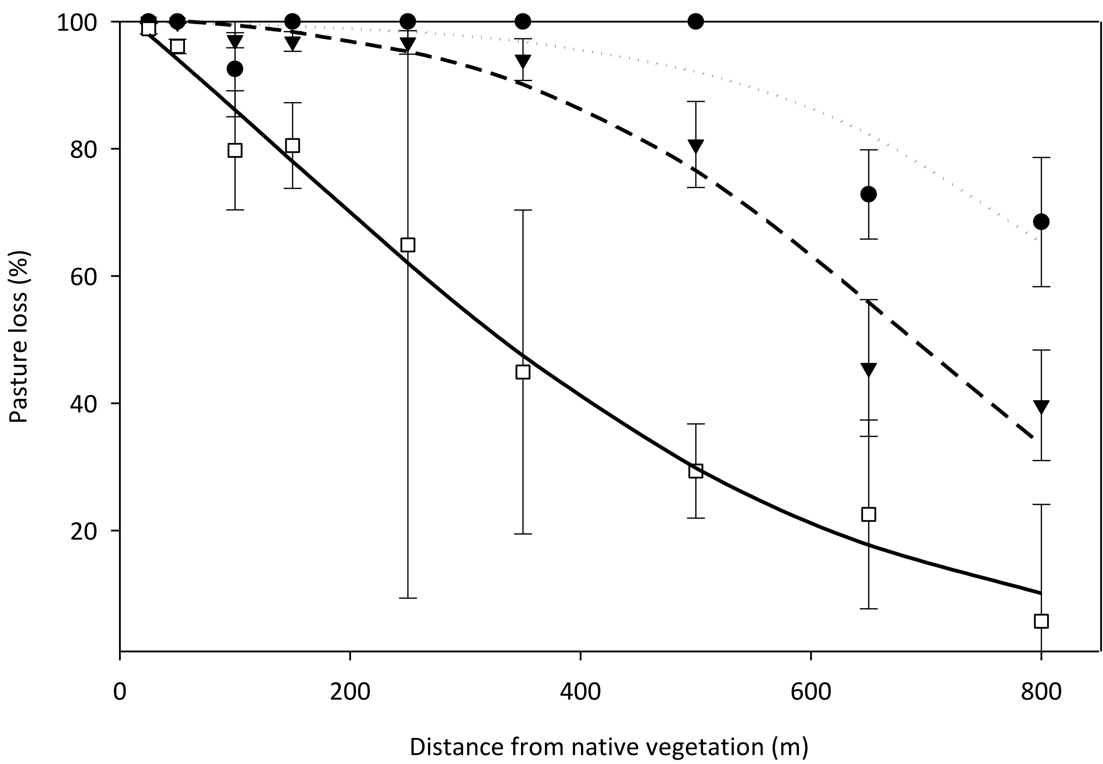


Figure 3.5 Proportion (%) of pasture lost to wildlife grazing in early winter (dotted, ●), early spring (short dashed, ▼), and late spring (solid line, □), at varying distances from the native vegetation edge during 2008. Error bars represent the standard error of the mean.

A similar pattern of pasture loss was observed in 2009 (Figure 3.6). Pasture losses were least during mid-spring, decreasing from 60% loss at 25 m to no impact at 800 m. Pasture loss in mid-summer was less than during late autumn and late winter at 25 m but all losses were similar across all seasons at 800 m. Pasture loss in late summer and autumn of 2010 remained high, above 90% for the first 450 m distance from native vegetation before decreasing sharply (Figure 3.7).

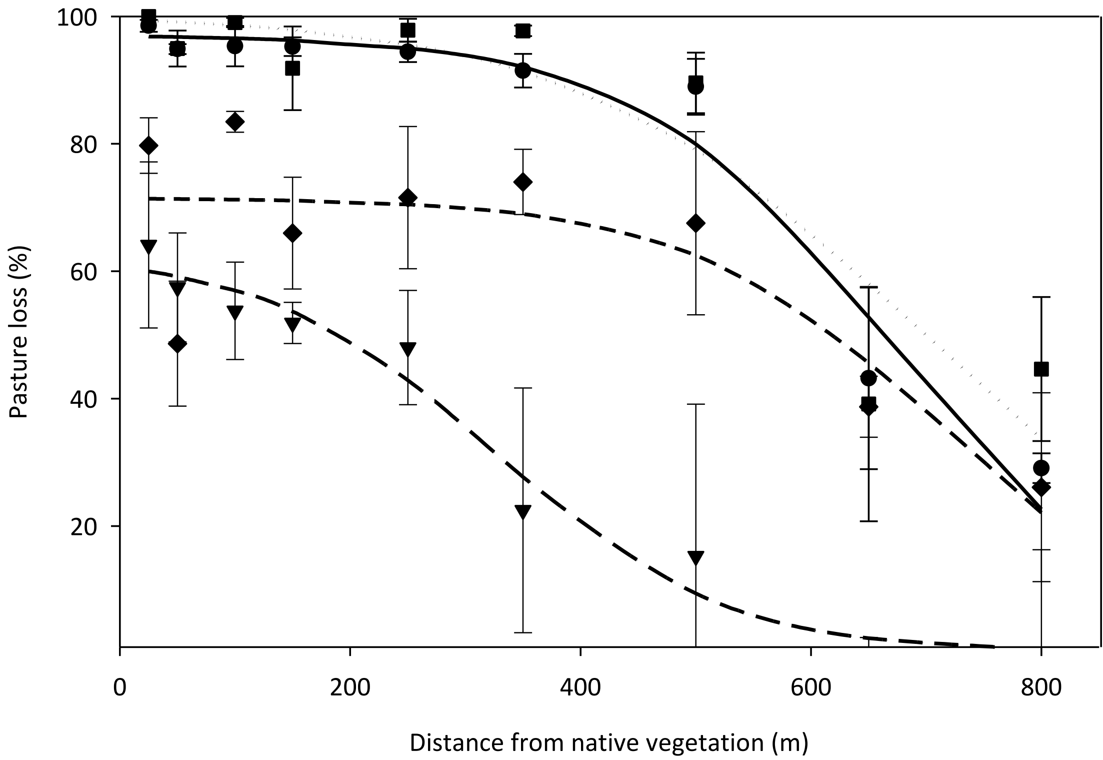


Figure 3.6 Proportion (%) of pasture lost to wildlife grazing in mid-summer (short dashed, ♦), late autumn (dotted, ■), late winter (solid line, ●), and mid-spring (long dashed, ▼), at varying distances from the native vegetation edge during 2009. Error bars represent the standard error of the mean.

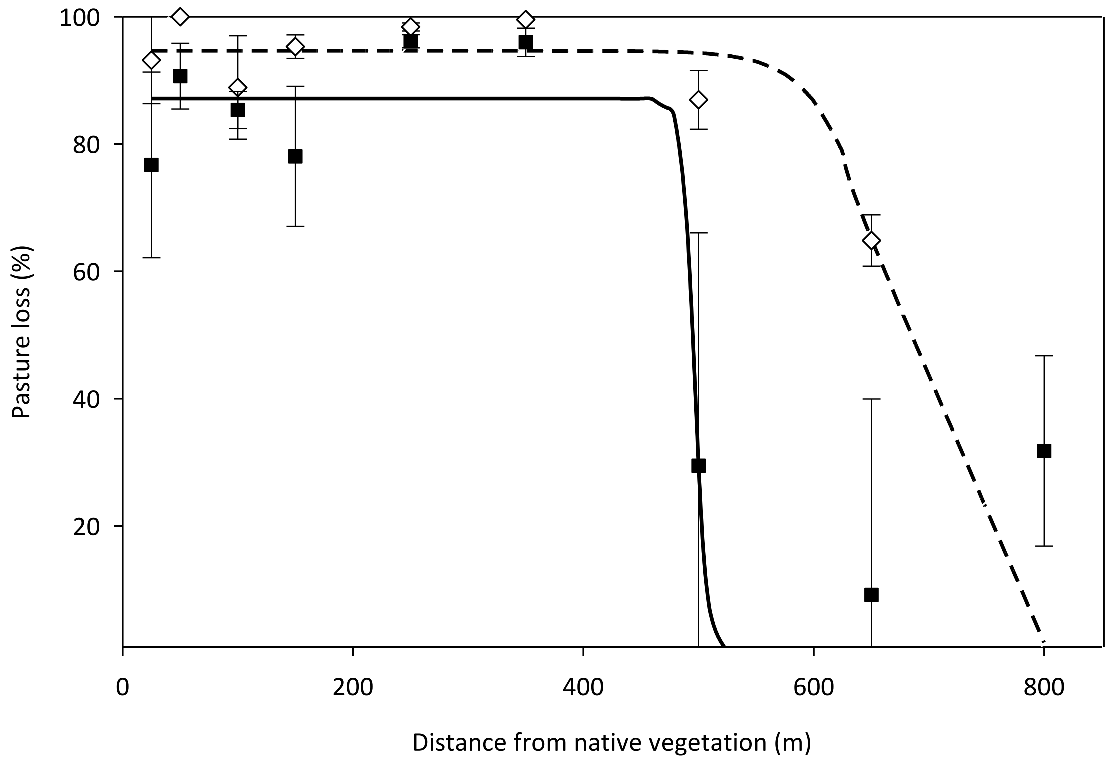


Figure 3.7 Proportion (%) of pasture lost to wildlife grazing in late summer (short dashed, ◊) and autumn (solid line, ■) at varying distances from the native vegetation edge during 2010. Error bars represent the standard error of the mean.

A summary of the logistic regression relationships for each harvest time is presented in Table 3.6. This includes the equation of the fitted pasture loss lines, the estimated distance at which 50% of pasture was lost and the average pasture loss per hectare over the first 800 m from the native vegetation. A relationship could not be found for the early summer 2009 harvest date, as small increases in production were found in the exposed plots compared with the enclosed plots. The distance from the native vegetation edge at which an estimated 50% pasture loss occurred varied considerably between harvest assessments, occurring closer to the native vegetation edge in spring than in any other season (Table 3.6). In spring 2008 and spring 2009 the distance from the native vegetation edge at which an estimated 50% pasture loss occurred was 332 m and 190 m, while in winter 2008 and winter 2009 the distance was 906 m and 663 m, respectively.

There was a significant ($t_9 = 3.80$, $P<0.01$) difference between the mean enclosed and mean exposed growth rates (Table 3.6).

Table 3.6 Summary of the logistic function L= a/1+EXP(-b-c*DIST) between pasture growth rates and distance from native vegetation edge for the period autumn 2008 to autumn 2010.

Time of harvest	Loss (L) equation	R^2	Distance at 50% reduction (m)
Early winter 08	$L = \frac{1.0043}{1} + EXP(-5.3778 - -0.00595 \times DIST)$	0.827	906
Early spring 08	$\boldsymbol{L = \frac{1.0239}{1} + EXP(-4.1012 - -0.00603 \times DIST)}$	0.959	688
Late spring 08	$\boldsymbol{L = \frac{1.5967}{1} + EXP(-0.5643 - -0.00407 \times DIST)}$	0.988	332
Mid summer 09	$\boldsymbol{L = \frac{0.715}{1} + EXP(-6.5055 - -0.00914 \times DIST)}$	0.714	619
Late autumn 09	$\boldsymbol{L = \frac{1.0049}{1} + EXP(-4.6209 - -0.00663 \times DIST)}$	0.852	698
Late winter 09	$\boldsymbol{L = \frac{0.9712}{1} + EXP(-6.0891 - -0.0091 \times DIST)}$	0.958	663
Mid spring 09	$\boldsymbol{L = \frac{0.6291}{1} + EXP(-3.2587 - -0.01 \times DIST)}$	0.969	190
Early Summer 09	NA.	NA.	NA.
Late summer 10	$\boldsymbol{L = \frac{0.9464}{1} + EXP(-21.5949 - -0.032 \times DIST)}$	0.997	671
Autumn 10	$\boldsymbol{L = \frac{0.9195}{1} + EXP(-22.2318 — 0.0439 \times DIST)}$	0.969	495

Table 3.7 Summary of pasture growth rates in enclosed and exposed plots in each measured season. The difference (reduction) between the enclosed and exposed plots is also provided.

Time of harvest	Mean kg DM/ha/day of enclosed	Mean kg DM/ha/day of exposed	Mean kg DM/ha/day reduction
Early winter 08	0.5 ±SE	0.0	0.5
Early spring 08	3.5 ±SE	1.0	2.5
Late spring 08	13.8	6.4	7.4
Mid- summer 09	12.6	4.7	7.9
Late autumn 09	2.5	0.4	2.1
Late winter 09	10.1	2.3	7.8
Mid spring 09	30.0	22.5	7.5
Early Summer 09	39.4	41.7	-2.2
Late summer 10	11.7	1.4	10.2
Autumn 10	15.5	3.0	12.5
Overall mean	**14.0**	**8.3**	**5.6****

Note: ** Denotes significance at $P<0.01$ based on a paired t-test.

3.3.2 Pasture growth rates

Mean pasture growth rates over the length of the trial period in the enclosed plots (14.0 kg DM/ha per day) were consistently higher than in the exposed plots (8.3 kg DM/ha per day). However, both growth rates were below the growth rates simulated by the SGS pasture growth model (23.18 kg DM/ha per day). Simulated growth rates were substantially higher than measured enclosed growth rates prior to April 2009, after which both growth rates were similar (Figure 3.8). The difference between the simulated growth rates and the observed growth rates was 21.1 kg DM/ha per day in October 2008. However, this difference in growth rates decreased to 10.5 kg DM/ha per day in September 2009. Growth rates of plots exposed to browsing decreased sharply in the summer of 2010 from above 30 kg DM/ha per day in November 2009 to effectively no measurable growth by January 2010 (Figure 3.9).

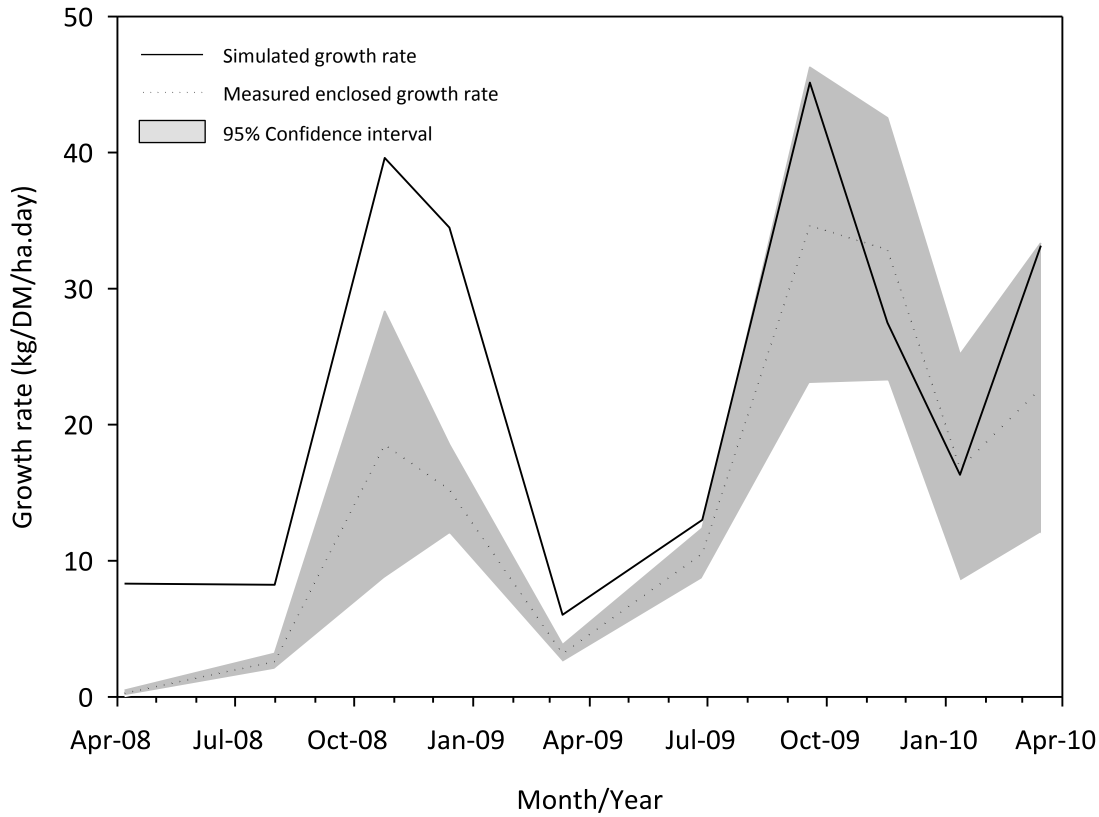


Figure 3.8 Measured pasture growth rates (kg DM/ha per day) of enclosed plots between April 2008 and April 2010. Simulated growth rates were obtained using the Sustainable Grazing Systems (SGS) pasture model and parameters (soil type and climate) specific for the Fosterville property.

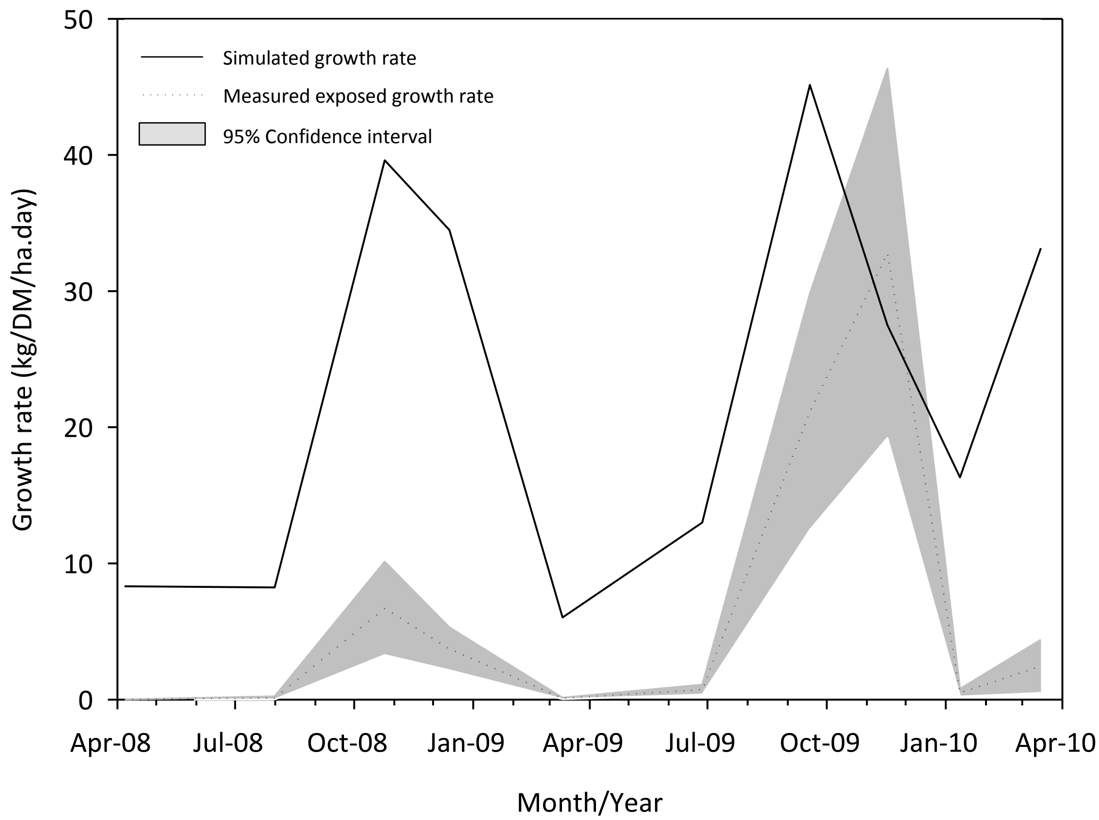


Figure 3.9 Measured pasture growth rates (kg DM/ha per day) of exposed plots between April 2008 and April 2010. Simulated growth rates were obtained using the Sustainable Grazing Systems (SGS) pasture model and parameters (soil type and climate) specific for the Fosterville property.

3.3.3 Wildlife feeding activity

Consistent with biomass results, wildlife (all species combined) feeding activity measurements showed decreases with distance from the native vegetation edge and decreases during spring (Figure 3.10). Feeding activity rates (as indicated by faecal pellets collected in g/day) decreased with distance from the native vegetation edge in all seasons. The feeding activity rates were below 0.2 g/day for all dates at 800 m. Rates were lowest for the spring 2009 collection, with minimal activity past 50 m from the native vegetation edge.

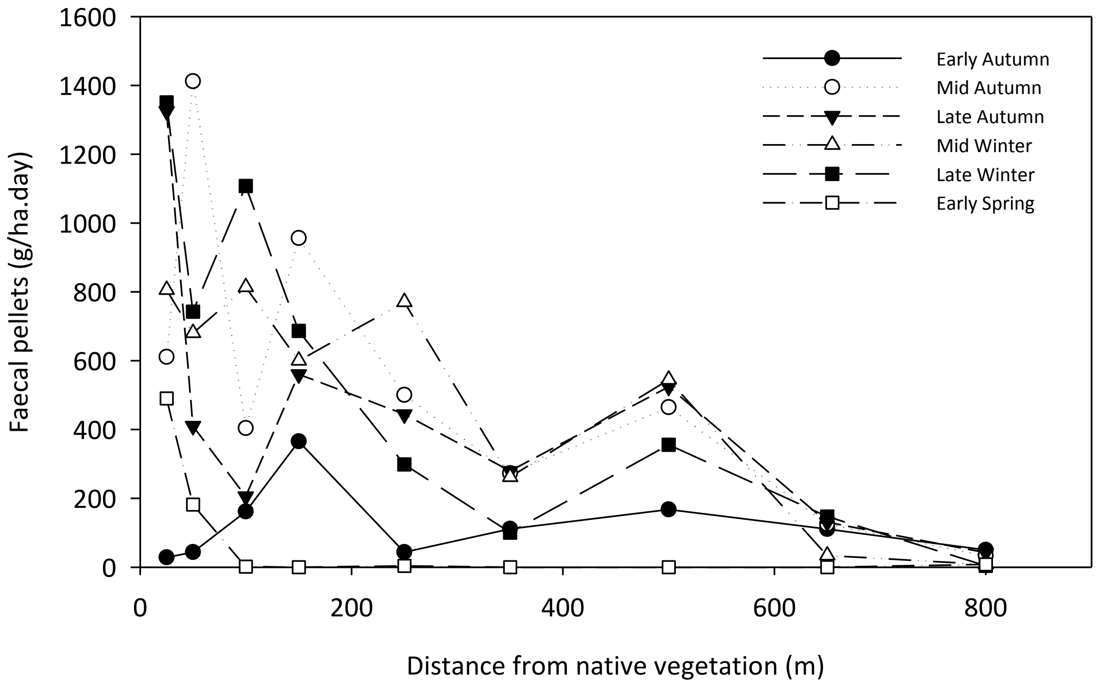


Figure 3.10 Feeding activity rates of all wildlife combined (as measured by faecal pellet weight) for early autumn, mid-autumn, late autumn, mid-winter, late winter, and early spring, at varying distances from native vegetation edge during 2009.

3.3.4 Correlation between pasture loss and feeding activity

There was a strong positive correlation between total pasture loss (enclosed – exposed) and total faecal weight of all grazing wildlife ($r = 0.893$, $P = 0.001$) (Table 3.8). Pasture loss increased with increasing faecal pellet weight and faecal pellet number (Figure 3.11 a, b). The proportion of pasture loss also increased with increasing pellet weight and pellet number (Figure 3.11 b, d). When total faecal weight was separated into individual wildlife species, pasture loss correlated most strongly with wallaby ($r = 0.872$, $P = 0.002$). The correlation between the proportion (enclosed/exposed) of pasture loss and total faecel weight was not as strong ($r = 0.754$, $P = 0.019$). Similarly, proportion pasture loss correlated most strongly with wallaby ($r = 0.637$, $P = 0.065$) compared with other wildlife species, but was not significant.

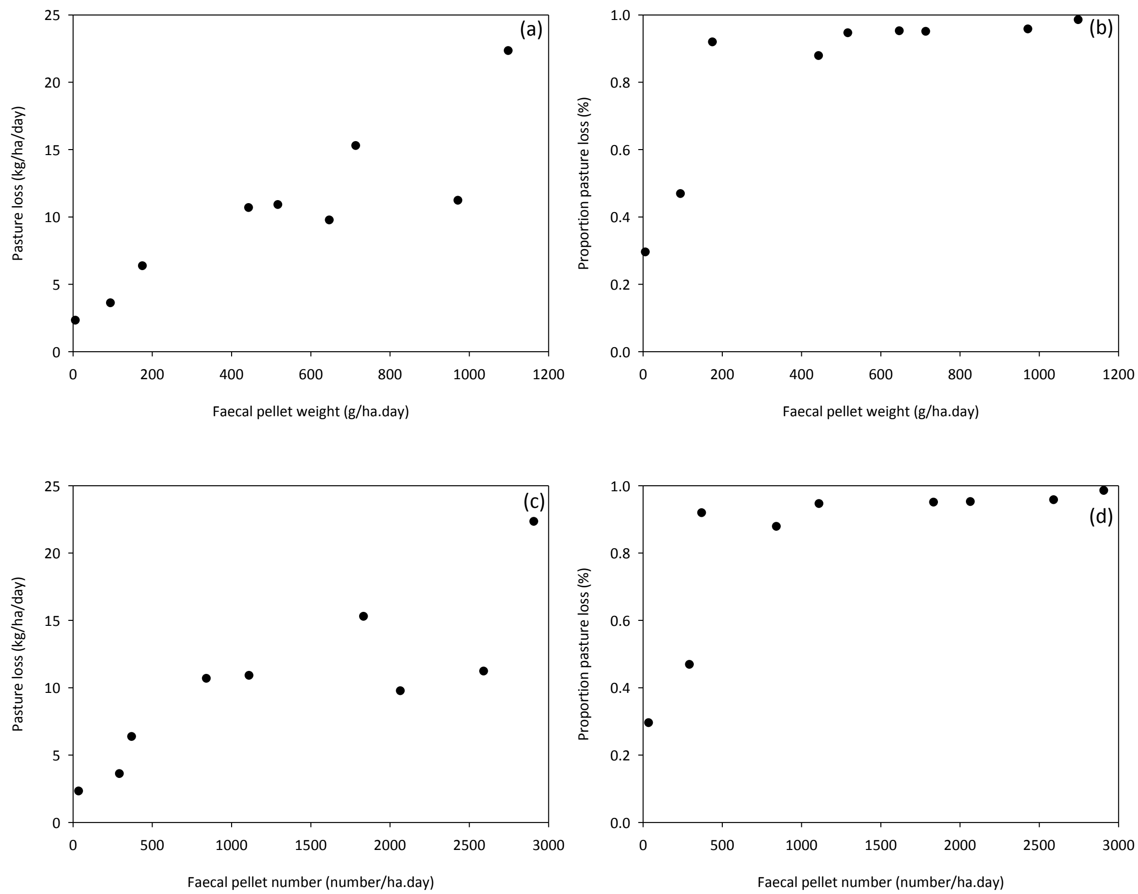


Figure 3.11 Scatterplot of; (a) measured pasture loss vs. faecal pellet weight, (b) proportion pasture loss vs. faecal pellet weight, (c) measured pasture loss vs. faecal pellet number, and (d) proportion pasture loss vs. faecal pellet number.

Table 3.8 Correlation coefficient table of measured total pasture loss and proportion pasture loss with measured faecal pellet weight and faecal pellet number. Pasture growth and faceal collection for the period 13th May to 11th August.

	Faecal weight		**Faecal number**	
	Total Loss	**Proportion loss**	**Total Loss**	**Proportion loss**
Kangaroo	0.035	0.273	0.011	0.258
Wallaby	0.872**	0.637	0.864**	0.631
Deer	0.058	0.209	0.058	0.209
Possum	0.399	0.454	0.447	0.479
Rabbit	0.010	0.353	0.007	0.331
TOTAL	0.893**	0.754*	0.842**	0.703*

Note: Symbols denote significance at * P<0.05, ** P<0.01.

3.4 Discussion

Reductions in pasture biomass associated with wildlife grazing were extremely variable and were influenced by the proximity to the native vegetation edge. Overall, significant differences in total accumulated biomass were recorded between enclosed and exposed plots up to 500 m from the native vegetation/pasture interface. Pasture loss was also affected by available pasture, which is also influenced by seasonal conditions. Significant differences between grazed and un-grazed plots were also observed by Robertson (1987). Large variations in pasture biomass were related to rainfall and grazing by kangaroos and sheep in a study by Robertson (1987) at Kinchega and Tandou in NSW. In that study, pasture biomass was closely associated with the amount of rainfall in the previous 6 months. Similarly, Fletcher (2006) found that pasture biomass and most closely associated with the amount of rainfall in the previous 12 months. Also, similar to the current study's findings, a 4-year study on the impact of western grey kangaroos (*Macropus fuliginosus*) in Western Australia found that crop losses varied significantly between 0% and 95% depending on crop type, season, and distance from adjacent woodland (Arnold *et al.* 1989).

Variation in the amount of pasture lost to grazing wildlife has been demonstrated by other trials within Tasmania. A short-term study by Donaghy and Tegg (2001) on a dairy farm in north-west Tasmania showed average DM loss was 34% on irrigated pasture and 21% on dryland pasture. Lower amounts of pasture loss in the study by Donaghy and Tegg (2001) may have been due to the location of the sites sampled. These appeared to be located at greater distances from the edge of native vegetation (not precisely defined, but inferred from a map). In addition, the availability of pasture at the site studied by Donaghy and Tegg (2001) was greater than that experienced at Fosterville in the current study. My study identified that the percentage reduction in DM increased with the advancement of the summer season. This was consistent with the study by Donaghy and Tegg (2001) and could be attributed to a decrease in availability of pasture. Statham and Rayner (1995) also showed substantial variation in the amount of DM loss, recording a loss ranging between 17 and 100% over 6 sites, with 5 of the 6 sites recording reductions in DM exceeding 40% DM. Distance from the edge of native vegetation was not considered as a treatment in either of these studies. In another trial, Statham (2000) found that carrying capacity losses of up to 37% could be attributed to wildlife grazing by wallabies. Kangaroos were thought to reduce winter wheat yields by between 10% and 17% in the Bungunya District of Southern Queensland in 1987 (Barnes and Hill 1992). These accounts indicate that the level of pasture or crop loss is extremely variable and influenced by a number of factors.

Measured pasture loss was greatest during summer, autumn and winter harvests and remained high (above 60% of available herbage) at greater distances from native vegetation (up to 500 m). The proportion of pasture loss was least during spring. Pasture growth rates are greatest during spring indicating that feed availability is likely to influence the proportion of pasture loss. Wildlife grazing on agricultural land during periods of low feed availability has been studied by others. Hill *et al.* (1988) found that kangaroos grazed on grain crops during winter and were in greater numbers during dry years.

Presenting the findings as the proportion loss as opposed to the amount lost to grazing needs to be carefully considered when comparing seasonal differences. Robertson (1987) showed that the amount of biomass grazed by kangaroos and sheep decreased during drought, but the proportion of the feed taken of the total feed available is likely to be increased. Likewise, Fletcher (2006) found that the amount of pasture biomass eaten by kangaroos increased with increasing availability of pasture biomass and that this relationship was linear. In addition, feed intake is influenced by quality in both sheep and kangaroos, with higher intakes at greater nitrogen contents (Foot and Romberg 1965; McIntosh 1966). With more pasture available to grazing wildlife in spring, it may be expected that the percentage pasture loss to wildlife grazing may decrease since the demand is likely to be heavily outweighed by supply of available pasture. This relationship was found to be the case in the current study. However, in a wider animal production context, sheep were lactating during spring and a greater amount of pasture was required to sustain growth of lambs. To complicate the matter further, in spring feed is more readily available in the native vegetation areas and the need to graze on sown pastures is reduced. In situations where stocking rates are increased to utilise more of the surplus pasture grown during the spring period, the impacts of wildlife grazing are likely to become more significant. In addition, farm productivity is likely to be impacted most during the autumn-winter period. At this time livestock and wildlife are likely to be competing for the same feed resource since the availability of pasture is lower than in spring. Newsome (1971) noted that competition between livestock and kangaroos would be greatest when green herbage becomes scarce.

At Fosterville sheep and wildlife are in competition for pastures. There is an overlap in diet between sheep and grazing wildlife since sheep are restricted to pastures and a range of wildlife species spend a significant period of time grazing these systems. The competition for feed resources at Fosterville may be more intense compared to other regions such as the Australian rangelands (Caughley 1987) where the feeding behaviour of these species have been examined. For example, the diets of sheep and kangaroos are reported to diverge during times of herbage shortages in rangelands, however competition for grasses at this time is at its strongest (Caughley 1987). Edwards *et al*. (1996)

discussed competitive interactions between kangaroos and sheep. Competition was intermittent, occurring only during a drought when food was depleted. At that time there was a large increase in the population density of kangaroos due to an influx of animals from neighbouring areas. Competition from kangaroos has been shown to reduce the live-weight of sheep (Edwards *et al.* 1996).

There are different ways to estimate the economic impost resulting from pasture loss to wildlife grazing. A simple technique to approximate the costs of wildlife grazing is to multiply the amount of pasture loss (t DM/ha) by the cost of purchasing an equivalent amount of feed. For example, the cost of importing one tonne of hay at Fosterville was estimated to be AUD $200 (S. Foster pers. comm.). For the current study in the year 2009, average pasture loss over the first 800 m from the native vegetation edge was 1.73 t DM/ha. This equated to an estimated economic impost of AUD $347/ha. Another method to approximate the costs of wildlife grazing is to calculate the opportunity cost due to reduced carrying capacity. The energy requirements of one DSE are 7.6 MJ/day (McLaren 1997; Victorian Government Department of Primary Industries 2005). This may equate to between 800 g and 1000 g DM/day for maintenance (W. Hunt pers. comm.). In this case, assuming that 900 g DM/day equates to 328.5 kg DM/annually, then 1.73 t DM/ha could potentially support an extra 5.3 DSE/ha. The net profit made by the top 20% of farms on one DSE for wool production in 2008 was estimated at AUD $13.81 (McEachern *et al.* 2009). Therefore, the estimated economic profit of the extra sheep that could be supported in this case is in excess of AUD $70/ha. However, the value of one DSE has increased significantly since 2008 (W. Hunt pers. comm.). The precise value will depend on commodity values and other on-farm factors.

While an estimate of pasture loss for the Fosterville property has been provided by the current study, pasture loss on other properties in the region is likely to be influenced by the amount of vegetation and the proximity of vegetation to improved pastures, season and feed availability and wildlife control measures implemented. Other studies have attempted to estimate the economic costs of grazing pests on livestock production. For instance, Barlow (1987) estimated the cost of each rabbit grazing to be NZ $2.10 due to reduced livestock stocking rates. In this case, rabbits had the greatest effect on sheep liveweight gain at higher sheep stocking rates (Barlow 1987). Furthemore, Fleming *et al*. (2002) showed that sheep liveweight, fat depth and fleece weights were negatively associated with rabbit density. This study highlight the considerable economic impost that grazing by wildlife has on the profits of graziers.

Control of wildlife during summer, autumn and winter, as opposed to spring, is likely to have a greater impact in protecting the comparatively smaller amount of pasture available for livestock and other production purposes. Control of wildlife through exclusion fencing is likely to increase pasture productivity in close proximity to native vegetation edges. A fence type consisting of ringlock topped with 3 wires was most effective in reducing production losses of crops in the study by Arnold *et al.* (1989). A 'wallaby wire' mesh fence was also successful in reducing pasture losses in the study by Statham (2000). Limiting grazing damage of wildlife is likely to extend the persistence and quality of introduced or improved pastures.

Differences between the growth rates of enclosed and exposed plots highlight the effect of wildlife grazing on production. The large initial (prior to April 2009) difference between measured enclosed and simulated growth rates shows a delay in the recovery of the previously exposed pasture. This indicates that wildlife grazing can have lasting effects on the production of improved pastures. Therefore, the exclusion of wildlife may not simply mean that pastures can return to predicted or expected growth rates. Pastures that have been heavily grazed by wildlife for extended periods may, depending on local circumstances, require re-sowing with improved pasture species to return pastures to acceptable or optimal growth rates. In a longer study, Neave and Tanton (1989) found significant differences in pasture height between exclusion cages that had been in place for greater than six years and matched plots grazed by kangaroos. Hence, the long term benefits of excluding wildlife from pastures may be greater than the measured short term benefits recorded here.

The rapid increase in growth rates of exposed plots during spring 2009, while later than the enclosed plots, shows the benefits of enhanced growing conditions. The rapid increase in growth rates may also be explained by compensatory growth of the pasture following a decline in grazing intensity. Growth rates may also have increased in response to high levels of soil nitrogen which may have mineralised and increased during the drought (Edwards and Herridge 2006). The rapid decrease in growth rates in the summer of 2010 of exposed plots, while enclosed plots were still growing at over 15 kg DM/ha per day, indicates the heavy grazing pressure of wildlife during this time. During the summer months available green herbage is in comparatively shorter supply than during spring, and as such is heavily grazed by wildlife. These results indicate that growth rates can quickly return to over 20 kg DM/ha per day during autumn if wildlife are controlled during the summer months.

Faecal pellet counts or collections have been used to examine habitat use and feeding activity of macropods (Caughley 1964; Hill 1978; Andrew and Lange 1986; Johnson *et al.* 1987; Banks 2001) and correlating feeding activity with pasture measurements seems appropriate (Southwell 1989). My study appears to be one of the first in Tasmania that has attempted to correlate pasture losses with faecal collections. In this case, pellet weights and pellet number were used as a measure of relative feeding activity, on the basis of Hill's (1978) findings that there is a strong association between faecal pellet output and feeding activity. In the current study, animals defecating in the survey areas were presumed to be feeding, given that the fixed survey areas were in pastures that provided no shelter. Johnson *et al.* (1987) noted that animals frequently defecate in the first 1-2 hours of feeding and rarely while inactive, indicating that fewer pellets would be found in a resting habitat and that the highest densities of pellets would be found in a feeding habitat. Some of the concerns raised by Bulinski and McArthur (2000a) were avoided in my study as the vegetation height was generally short, and pademelons were absent.

Habitat usage by grey kangaroos was studied by Hill (1981b), who found that generally, environments that provided a balance between food reserves and cover were well populated. Hill (1981b) explains that kangaroos will remain in areas that provide cover while feed is available, but periods of feed shortages (drought) forces kangaroos to move further from cover in search of food. Newsome (1971) also noted thatonce herbage supply diminished red kangaroos (*Macropus rufus*) moved from woodlands into open plains and creek flood-outs in search of green herbage. Studies by Banks (2001) explain the importance of cover for predator avoidance, with kangaroos spending more time closer to cover when the risk of predation is high. Furthermore, studies on red-necked pademelons (*Thylogale theties*) by Wahungu *et al.* (2001) showed that this species preferred to graze close to cover, and when grazing further from cover increased group size in an attempt to reduce the risk of predation. As such the trade-off between accessing food and remaining in areas that are relatively safer from predators remains an ongoing paradigm for wildlife. The diurnal patterns of resting in cover during the day and grazing in more open grass covered areas during the night is a strategy that is common among macropods and was observed in the current study for not only wallaby and kangaroo, but also for deer and possum.

The results from the current study of wildlife feeding activity measurements are consistent with the observations by Hill (1981b) showing that wildlife will move further from cover as feed availability within or close to cover diminishes; and consistent with findings in studies by While and McArthur (2005; 2006) showing that wildlife prefers to graze close to native vegetation edges. Both the weight of faecal pellets and the percentage reduction in pasture production decreased with distance from the native vegetation edge, and varied seasonally. While and McArthur (2006) found similar results in a young forestry plantation habitat, with distance from cover influencing artificial food-patch depletion by wallabies and pademelons. While and McArthur (2005) reported that faecal pellets of pademelon decreased with distance from cover while pellets of wallaby increased with distance from cover. This may be due to interspecific competition between pademelons and wallabies. These results contrast with the decrease in pellets with distance in the current study; however this may be related to the combination of grazing species present. Studies by Southwell (1987) showed that eastern grey kangaroo numbers increased with distance from forest edge during the day while red-necked wallabies (*Macropus rufogriseus banksianus*) remained closer to the forest edge. Similarly, in the current study kangaroos were observed to be the first to venture from cover to graze and ventured further into the pasture than wallabies (see section 2.12).

In a grazing systems context, overgrazing of pastures and competition for food resources is likely to occur where both sheep and wildlife graze. In this instance it is within the first few hundred metres from the native vegetation/pasture interface. Interestingly, Terpstra and Wilson (1989) found that sheep had little preference for specific vegetation types and grazed randomly and unevenly in relation to feed availability, while kangaroos showed a preference for areas with some form of tree or vegetation cover. In instances where sheep and kangaroos grazed together the damage to the pastures was likely to be more severe (Terpstra and Wilson 1989). This highlights the challenges landowners face in managing and grazing pastures sustainably. Overgrazing leads to degradation of the pasture through a reduction in perennial pasture species and an increased chance of soil erosion leading to productivity decline.

A relationship was demonstrated between grazing damage and feeding activity by Bennett's wallaby. While, the weight and number of faecal pellets was assessed as an index of feeding activity for all species in this study, a corelation was only established for wallaby (Table 3.8). This finding may indicate that this species was having the greatest impact on pasture loss. Pasture loss appears correlated with both faecal pellet weight and faecal pellet number. An assessment of faecal pellets within a survey area could provide an indication of pasture loss. The weight and number of faecal pellets required to have a significant pasture loss was low. For example, just 175 g/ha/day or 370 pellets were linked to a 92% reduction in DM. Greater weights and number of pellets beyond this resulted in near 100% reduction in pasture. The addition of weight past 600 g/ha/day and number past 1500/day had little further impact on proportion of pasture lost. Assuming that wildlife were still grazing within the area, this indicates that wildlife were grazing pasture to a level below that which was assessed (<20 mm). For a considerable decrease in the proportion of pasture loss to occur, wildlife control leaving very few animals would be required.

3.5 Conclusions

In conclusion, this chapter had several aims. The first aim was to quantitatively investigate wildlife grazing on established pastures by testing for correlations between pasture loss to grazing and distance from native vegetation, and season. The second and related aim was to quantify how grazing pasture loss varies both spatially and temporally in perennial pastures. My research established correlations between pasture loss from wildlife grazing and distance from native vegetation. On average, the greatest loss of pastures to wildlife grazing occurred close to native vegetation and declined with distance from native vegetation. Based on direct and indirect observations, different species of wildlife were involved in grazing of pastures at distance to native vegetation, and these differences were consistent with the known foraging behaviour of the wildlife present during the study period.

Correlations were established between the extent of pasture loss to wildlife grazing and season. Seasonal variation in pasture production was related to the proportional loss of pastures to grazing over time. Measured pasture loss was greatest during summer, autumn and winter harvests and least during spring. Lower numbers of wildlife fed on pasture during spring and I suggested this pattern occurred because of the higher herbage availability to wildlife within native vegetation areas at this time.

The third aim was to quantify the effectiveness of the Sustainable Grazing Systems (SGS) pasture growth model to predict pasture growth on the Fosterville property, and examine the residual effects of wildlife grazing. The pasture growth rates simulated by the SGS pasture model were initially higher than the measured growth rates in the study. The residual effects of heavy grazing by wildlife and drought may have been underestimated in the simulation. This may suggest that the SGS pasture model may not provide realistic simulated growth rates in this region when pastures are stressed by heavy grazing and drought. The SGS pasture model provided simulated growth rates that were more comparable with measured growth rates in the second half of the study period when Fosterville received more average rainfall, and the residual effects of heavy grazing may have diminished.

The fourth aim of the chapter was to test if a relationship exists between observed grazing damage and an index of feeding activity of native herbivores. My research demonstrated a relationship between observed grazing damage and an index of feeding activity (based on the weight and number of faecal pellets per unit area). Changes in the relative abundance and foraging behaviour of wildlife species were consistent with the observed loss in pasture production, and the amount of pasture lost to grazing was inversely related to distance from the edge of native vegetation. Both the weight of faecal pellets and the percentage reduction in pasture production decreased with distance from the edge of native vegetation.

The final aim of the chapter was to quantify and evaluate the economic costs of wildlife grazing in the Midlands region of Tasmania. The empirical results of my research enabled a quantification and evaluation of the economic costs of pasture loss to wildlife grazing in the Midlands region. I calculated that the average pasture loss over the first 800 m from the edge of native vegetation at Fosterville during 2009 was valued at AUD \$366 ha^{-1}. I calculated that the pasture lost to wildlife grazing equated to a reduction in livestock carrying-capacity at Fosterville of 6.5 DSE ha^{-1}. Based on these estimates, the loss of pastures to wildlife grazing in comparable areas of the Midlands is likely to be significant.

The results presented in this Chapter indicate some of the direct effects of grazing wildlife on the production of improved pastures. However, indirect and less obvious effects of wildlife grazing may be significant as well. For example the impact of continual overgrazing of pastures by wildlife may affect the botanical composition of the pasture and influence the amount of ground cover. Chapter 4 considers some of these issues.

Chapter 4: Effects of wildlife grazing on ground cover and plant species composition of an established perennial pasture

4.1 Introduction

The effects of wildlife grazing on plant species composition has been studied in a numerous of regions of the world. For example, exclusion of reindeer (*Rangifer tarandus*) from the forest-tundra vegetation in Finnish Lapland resulted in increased vegetation cover and varying responses among plant groups (Pajunen *et al.* 2008). Studies on the effects of grazing intensity of Tibetan sheep (*Ovis ammon*) on shrubby meadows on the Qinghai-Tibet Plateau showed differences in species composition with a decline in palatable species, litter, and vegetation height at high intensities, although species richness was not affected (Zhou *et al.* 2006). Declines in palatable species and species richness were recorded close to overnight stock enclosures used for goats and sheep (likely *Capra hircus* and *Ovis aries*) in Richtersveld National Park, South Africa (Hendricks *et al.* 2005). Vegetation height and cover of clover was reduced and stability of the soil decreased by cattle (*Bos taurus*) grazing on rehabilitated forest landings in British Columbia (Krzic *et al.* 2006). Heavy grazing by livestock on rangeland vegetation resulted in the dominance of a few species and a reduction of some grazing sensitive species in Niger (Hiernaux 1998). Species richness and species frequency generally declined closer to water points where grazing intensity by livestock was highest in rangelands in Australia (Landsberg *et al.* 2003).

Despite anecdotal reports such as reductions in the legume components of pastures, and reductions in ground cover, there is little known about the effect of wildlife grazing on the botanical composition and ground cover of improved pastures in Tasmania. Changes in composition of grasslands caused directly and indirectly by grazing kangaroos have been studied on mainland Australia. Studies by Neave and Tanton (1989) at the Tidbinbilla Nature Reserve in the Australian Capital Territory showed decreased percentage cover of *Themeda australis*, *Glycine clandestina*, *Aira caryophyllea* and *Haloragis tetragyna* in exclusion cages grazed by grey kangaroo *Macropus giganteus.* In contrast, other species like *Bromus molliformis*, *Trifolium sp*., and *Hypochoeris radicata* increased in percentage cover.

The composition of plant species within a pasture has a direct influence on the level of pasture production. Improved pastures in south-eastern Australia are based on exotic grasses and legumes (Blair 1997). Cultivars have been developed to have characteristics such as: high DM production, high nutritional value, and the ability to withstand repeated grazing and grow at times of the year when feed is in short supply. To maximise production of pasture it is important to maintain the content of these improved plants in the sward while restricting plants with less desirable traits.

From a production prospective, undesirable plants can include annual grasses and broadleaf weeds that are less productive, have shorter growing seasons, have seed heads which can contaminate wool and which may be toxic. The botanical composition of the pasture can be changed by strategic grazing (Lodge and Whalley 1985). It is important to maintain a balance between grasses and legumes as grasses can out-compete legumes if pasture is understocked with a consequent reduction in nitrogen supply to the sward (Hill *et al.* 2004). By managing grazing appropriately, in this instance diversity (grass + legume) can be encouraged in the pasture. This finding appears consistent with the theory that predators may enhance the coexistence of competitors in a system (Caswell 1978). In this case, plant diversity may be enhanced in the presence of grazing wildlife.

Persistence of an improved pasture refers to the ability of a pasture to remain productive long-term (20+ years). As such, the botanical composition is directly linked with the persistence of a pasture. A decrease in the proportion of desirable improved plant species results in decreased production and over time the need for pasture improvement by re-establishment. The profitability of perennial pasture grazing is partly linked to this cycle of establishing, grazing and re-establishing new pastures over time. The greater the period of time between establishment and re-establishing the lower the cost of production.

Persistence can also refer to survival of individual species within a pasture. For example, a deep rooting species such as phalaris may have greater persistence in low rainfall areas than a shallow rooted species, while the prostrate growing characteristic of bent grass (*Agrostis spp.*) better equips it to persist under heavy grazing than the erect growth habit of prairie grass (*Bromus willdenowii*). Therefore, before establishment it is important to match the pasture species to the environment and management conditions under which it is to be grown.

Several factors can influence the species composition of a pasture including; soil fertility and the applications of fertilisers (Hill *et al.* 2004), rainfall and extended periods of drought (Hutchinson *et al.* 1995; Jones *et al.* 1995), damage from insect pests (McQuillan *et al.* 2007), and grazing (Jones 1992; McIvor and Gardener 1995; Kemp 1999; Hill *et al.* 2004). In Tasmania, insect pasture pests like corbie larvae (*Oncpera intricate*) feed on improved grasses and legumes and generally avoid feeding on broadleaf weeds (McQuillan *et al.* 2007). This causes a change in composition from improved species to weeds while leaving bare patches in the pasture open to invasion from annual grasses. Maintaining ground cover of sown pasture species is vital in preventing the ingress of weedy annual grass species and broadleaf weeds. Reducing ground cover by overgrazing has also been shown to induce erosion (Scott and Kirkpatrick 2008).

Livestock grazing at high stocking rates can change the species composition and amount of ground cover in a pasture. Generally, as a result of overgrazing, perennial grass species decline, the amount of bare ground increases, allowing annual grasses and broadleaf weeds germinate and fill in the void. Increasing the stocking rate beyond the capacity of the pasture to supply feed is one form of overgrazing that can lead to an increase in the amount of bare ground (Kemp 1999; McGregor 2010). The effects of rabbit grazing on the tall tussock grasslands of Macquarie Island included an increase in the amount of bare ground, erosion and landslips (Scott and Kirkpatrick 2008).

Grazing can be used as a tool to modify the species composition of a pasture. This can be achieved by selectively grazing the pasture at different life stages of the species present within the pastures as there is seasonality in the way individual species grow. For example, resting desirable species when they are most sensitive to grazing, while grazing the pasture when undesirable species are most sensitive to grazing can positively alter the composition (Kemp 1999). Spelling of the pasture from grazing during seed set can ensure desirable plant species remain persistent (Campbell *et al.* 1987). In contrast, grazing improved grasses such as phalaris during early establishment can have a negative effect on species composition (Campbell *et al.* 1987). Pastures require time to recover following grazing to allow new shoots to grow and energy reserves to be replenished. Wildlife grazing being continuous and uncontrolled may therefore have a negative effect on species composition.

Few studies have investigated the effects of wildlife grazing on production of pastures, and thus there is limited knowledge on the impact of wildlife grazing regarding improved pasture species composition. This information is required to assess the ongoing persistence and productivity of pastures growing in close proximity to native vegetation and plantation edges. This chapters aims to:

- Quantify the effects of wildlife grazing on pasture species composition and ground cover in pastures;
- Test for correlations between wildlife grazing and a decline in the composition of improved pasture species over time;
- Test if wildlife grazing may contribute to an increase in the cover of less desirable grasses and broadleaf weeds; and
- Quantitatively determine if protecting pastures from wildlife grazing can increase ground cover and reduce the susceptibility of pastures to erosion.

4.2 Methods

A full description of the experimental site and experimental design is given in section 3.2.1 and 3.2.2. Exclusion cages were erected at varying distances from a native vegetation/pasture interface at 25 m, 50 m, 100 m, 150 m, 250 m, 350 m, 500 m, 650 m and 800 m from the vegetation edge. Each distance boundary contained 4 caged replicates with paired ‘exposed’ plots within 2 m of the cage. Comparisons of species composition were made between the enclosed (caged) and exposed plots at each distance boundary.

4.2.1 Initial pasture survey

The pasture was assessed initially in April 2008 using the visual estimate method (outlined in section 4.2.2) to establish the species composition. The pasture varied considerably with distance from the edge of native vegetation. It was established that boundaries at 25 m and 50 m were dominated by annual grasses and broadleaf weeds. Boundaries at 100 m, 150 m, 250 m, 350 m, and 500 m were dominated by perennial ryegrass, barley grass, subterranean clover and cocksfoot. Boundaries at 650 m and 800 m were dominated by phalaris with small amounts of ryegrass, cocksfoot and subterranean clover.

To assist with the assessment and comparison of results, 3 zones of like composition were designated. The zones were based on a combination of distance from the native vegetation and the botanical composition of each of the boundaries. Zone 1 consisted of distance boundaries 25 m and 50 m, being very close to the native vegetation and dominated by annual grasses. Zone 2 encompassed boundaries 100 m, 150 m, 250 m, 350 m, and 500 being an intermediate distance from the native vegetation and containing a mixture of plants species including introduced perennial grasses and annual grasses. Zone 3 included boundaries 650 m and 800 m and these boundaries were dominated by phalaris.

4.2.2 Assessment of plant species composition

Species composition

The pasture species composition of the paired plots was assessed using two methods. The first was by the hand separation of harvested samples conducted in the laboratory prior to drying. Accumulated pasture was harvested (as detailed in section 3.2.3) and plant parts of whole samples were separated based on species. Each individual species was then oven dried at 60°C for 48 hours (Sanford *et al.* 2003), weighed and the percentage yield contribution was calculated on a dry weight basis. Broadleaf weeds were omitted from the analysis as broadleaf sprays were used intermittently to remove thistles. Larger thistles were also hand removed to make it easier for the hand separation of the other pasture species. Botanical composition survey dates are presented in Table 4.1.

Table 4.1 Summary of botanical composition survey dates

Procedure	Date
Hand separation	2-6-2008, 29-9-2008, 19-11-2008, 7-1-2009, 13-5-2009, 11-8-2009, 26-10-2009, 9-12-2009, 15-2-2010, and 13-4-2010
Visual cover estimates	24-4-2008, 2-6-2008, 29-9-2008, 19-11-2008, 7-1-2009, 13-5-2009, 28-7-2009, 22-10-2009, 9-12-2009, 15-2-2010, and 13-4-2010

The second method used to assess species composition was a visual estimate of the amount of canopy each pasture species occupied when projected against the soil. This was recorded as a percentage along with the percentage of bare ground/residue remaining. This method was adapted from a method for canopy cover outlined in t' Mannetje (2000b).

For convenience of data presentation, phalaris, ryegrass and cocksfoot (preferred by graziers for production) were grouped and termed 'desirable'. Similarly weedy annual grasses and broadleaf weeds (less preferred for production) were termed 'undesirable'.

4.2.3 Data analysis

Paired t-tests were performed using the statistical analysis package SPSS (IBM SPSS Inc. 2010) version 19.0.0 to compare means of botanical composition between enclosed and exposed to wildlife grazing treatments.

4.3 Results

4.3.1 Spatial variability of plant species composition

The results for the plant species composition that were obtained from the initial survey of botanical composition, hand separation and visual cover estimates were consistent and indicated that the pasture could be defined as three zones based on plant species. Zone 1 was dominated by annual grasses, Zone 2 contained a mixture of improved perennials, subterranean clover and annual grasses, and Zone 3 was dominated by phalaris (Figure 4.1). The amount of bare ground at 25 m was 67% and was higher than all other boundaries which ranged between 44 and 54% (Figure 4.2). There was greater broadleaf weed cover at 25 m (5.5%) and 50 m (9.3%) than at all other boundaries.

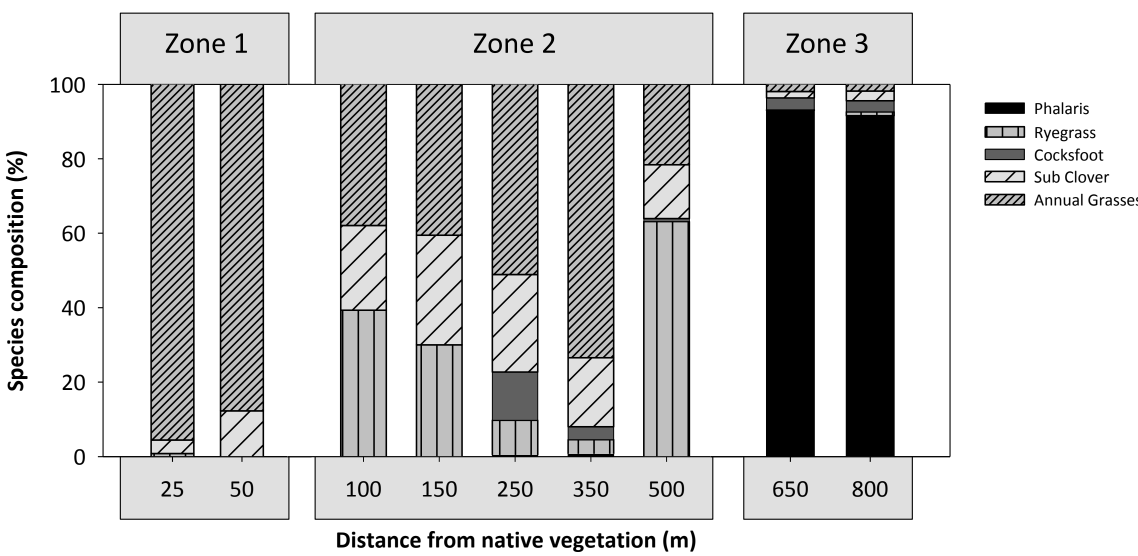


Figure 4.1 Species composition determined on a dry weight basis. Trial establishment was 11th February 2008 and samples were harvested between 2nd June 2008 and 13th April 2010. The pasture is segmented into 3 zones based on dominant pasture grass. Data from exposed plots only are presented and are an average of each individual plant species over the 2 years of the experiment.

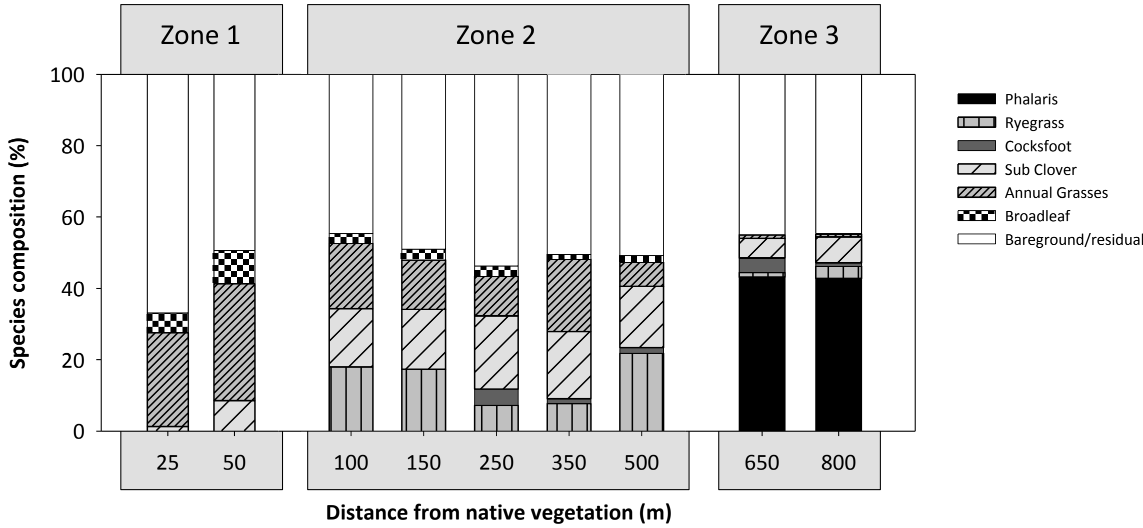


Figure 4.2 Species composition by visual estimation of cover (%). Pasture was surveyed between 28th March 2008 and 13th April 2010. The pasture is broken up into 3 zones based on dominant pasture grass. Data from exposed plots only are presented.

4.3.2 Seasonal variability of plant species composition

The DM yield of individual pasture species varied among zones, seasons and enclosed and exposed treatments (Figure 4.3). Above ground biomass accumulation was greatest during the spring/early summer period in all zones. Dry matter yield for all species was greater in the 2009 than the 2008 growing season. Seasonal changes in the botanical compositions, as determined by the hand separation method, are given in Figure 4.4. Seasonal changes in the amount of bare ground/residue and amount of broadleaf weed cover, as determined by the visual assessment method are given in Figure 4.5.

Zone 1

In Zone 1, annual grasses dominated mean (± standard error (SEM)) total accumulated DM yield (kg DM/ha) and was significantly ($t_7 = 8.20$; $P<0.001$) higher in enclosed (10,734 ± 990 kg DM/ha) than in exposed (3,316 ± 458 kg DM/ha) plots (Figure 4.3). Yield of annual grasses on each of the harvest dates was significantly ($P<0.05$) higher in enclosed than exposed plots except on 2/6/2008 and 9/12/2009. In addition, greater amounts of subterranean clover were harvested in the enclosed (730 ± 204 kg DM/ha) than exposed (272 ± 93kg DM/ha) plots, but this was not significant ($t_7 = 1.809$; $P>0.05$).

In contrast to annual grasses, yield of subterranean clover was only significantly (t_7 = 2.41; P<0.05) higher in enclosed than exposed plots on the final harvest date.

The percentage of subterranean clover in the sward increased in enclosed plots in April 2010 (Figure 4.4). The contribution of ryegrass in the enclosed plots increased particularly towards the end of the two year trial, from 0% in summer 2009 to 13% in summer 2010 (Figure 4.4). Subterranean clover DM also increased in the summer of 2010 in enclosed plots. There was a very low proportion of 'desirable' grasses in terms of cover expressed as a percentage (Figure 4.5a), while the cover of 'undesirable' plants was higher in enclosed than exposed plots on most harvest dates (Figure 4.5g). There was greater cover of subterranean clover in enclosed than exposed plots in 2010 (Figure 4.5d). Bare ground cover was generally higher in the exposed plots, and particularly so in 2008 and early 2009, but was similar to enclosed plots in the summer of 2010 (Figure 4.5j).

Zone 2

In contrast to Zone 1, mean (± SEM) total accumulated DM yield (kg DM/ha) in Zone 2 contained a combination of species including annual grasses, ryegrass, cocksfoot and subterranean clover (Figure 4.3). Production of cocksfoot increased in enclosed treatments over the trial period (Figure 4.3).There was a significantly (t_{19} =4.05; P= 0.001) higher total yield of cocksfoot in enclosed (3,722 ± 908 kg DM/ha) than exposed (105 ± 50 kg DM/ha) plots. Likewise there was significantly (t_{19} = 2.17; P<0.05) and (t_{19} = 2.33; P<0.05) higher total yield of ryegrass (2,683 ± 653 compared with 1,315 ± 483 kg DM/ha) and annual grasses (2,612 ± 409 compared with 1,555 ± 247) in enclosed than exposed plots respectively.

In contrast, there was higher total yield of subterranean clover in exposed (781 ± 117 kg DM/ha) than enclosed (651 ± 169 kg DM/ha) plots, but this was not significant (t_{19} = 0.71; P = 0.490). The difference between enclosed and exposed plots was particularly noticeable during spring and summer 2009 when the proportion of subterranean clover contributing to the harvest yield increased in exposed plots (Figure 4.4).

The proportion cover of desirable grasses in Zone 2 was greater in enclosed plots than in exposed plots on all harvest dates (Figure 4.5b). The proportion cover of subterranean clover was greater in exposed plots during spring 2009, reflecting the higher yield reported earlier (Figure 4.5e). There was no difference in the cover of undesirable plant cover (Figure 4.5h), but the amount of bare ground was generally greater in exposed than enclosed plots (Figure 4.5k).

Zone 3

The total DM yield in Zone 3 was dominated by phalaris with no significant ($t_7 = 1.56$; $P>0.05$) difference between enclosed (5765 ± 321 kg DM/ha) and exposed (5272 ± 349 kg DM/ha) treatments (Figure 4.3). There were small significant differences between yield of phalaris in enclosed and exposed plots on some of the harvest dates. For example, yield in enclosed plots was significantly ($t_7 = 3.28$; $P<0.05$) higher than exposed plot on 11-8-2008. Minor differences in the proportion of ryegrass and cocksfoot in the yield of enclosed and exposed plots were recorded in Zone 3 (Figure 4.4). The proportion cover of desirable plants, subterranean clover, undesirable plants, and the amount of bare ground was comparable in enclosed and exposed plots (Figure 4.5c, f, i and l).

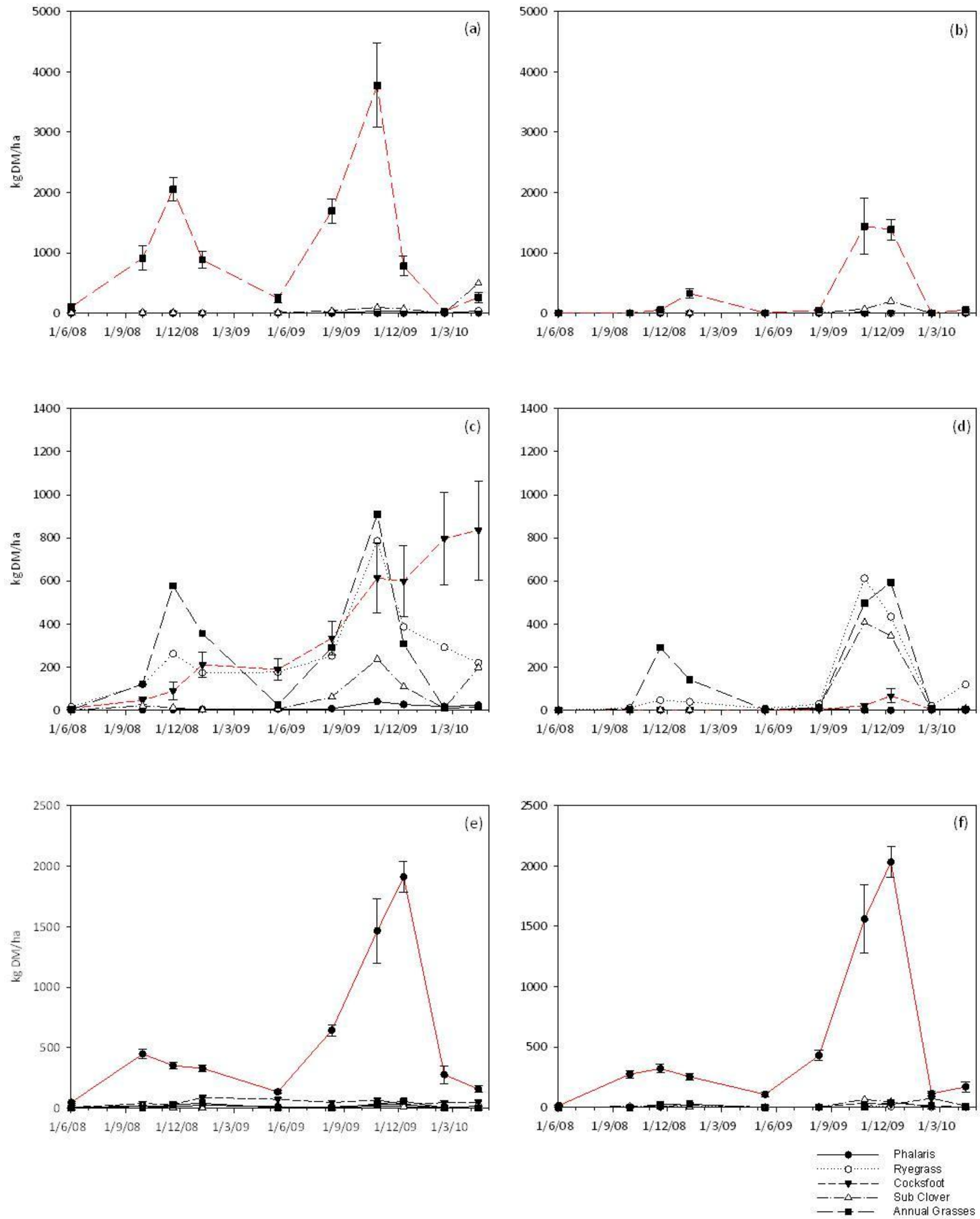


Figure 4.3 Dry matter (kg DM/ha) yield of individual pasture species from harvests between June 2008 and April 2010 from Zone 1 (a, b), Zone 2 (c, d) and Zone 3 (e, f) under enclosed (a, c & e) and exposed (b, d & f) treatments. Red lines indicate species of the most interest and include error bars that represent the standard error of the mean.

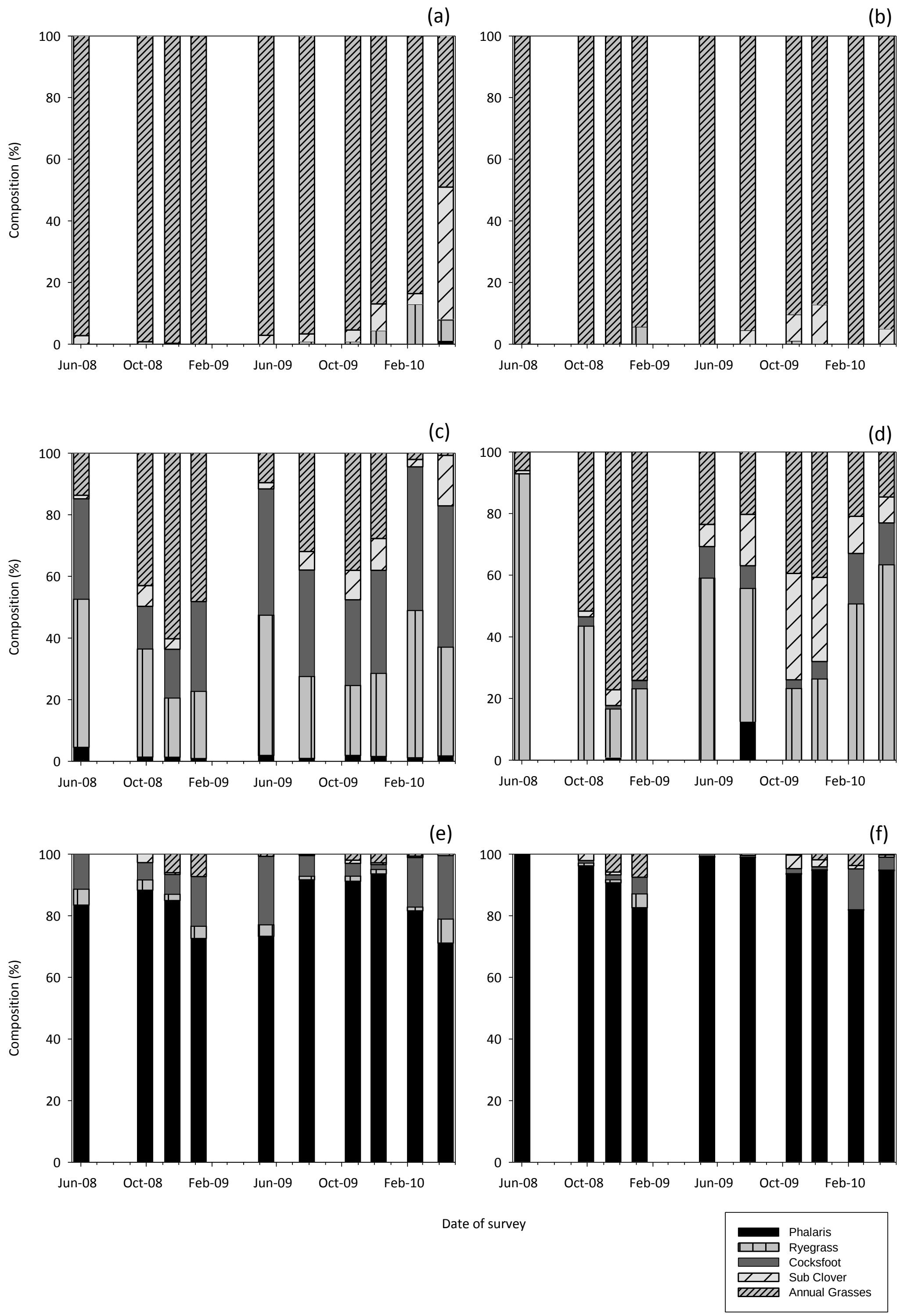


Figure 4.4 Species composition (%) by dry matter yield of individual pasture species from harvests between June 2008 and April 2010 from Zone 1 (a, b), Zone 2 (c, d) and Zone 3 (e, f) under enclosed (a, c & e) and exposed (b, d & f) treatments.

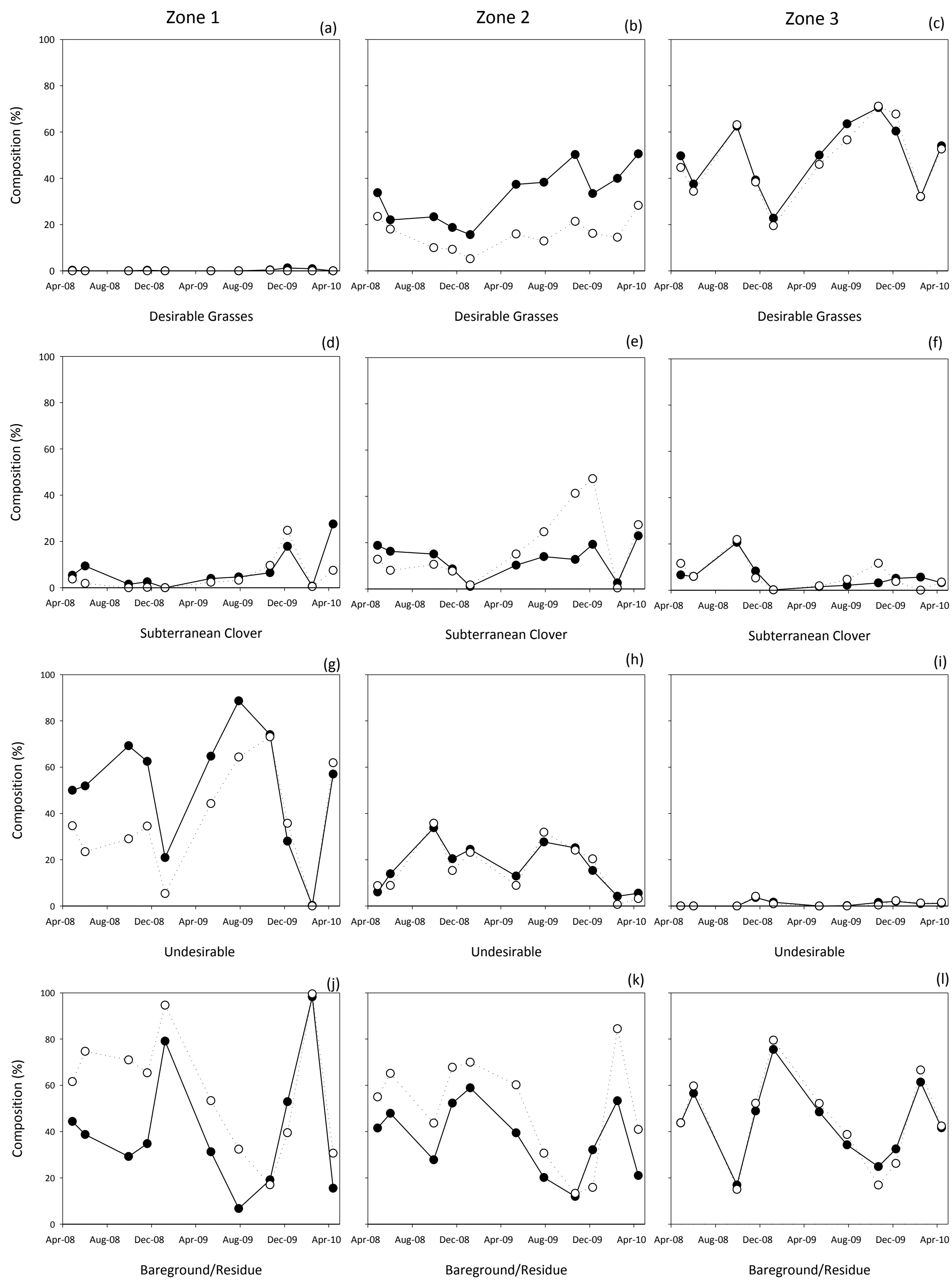


Figure 4.5 Species composition (%) by cover of desirable grasses, subterranean clover, undesirable plants and bare ground/residue from surveys between March 2008 and April 2010. Enclosed treatments are represented by filled markers and exposed treatments by unfilled markers in zone 1 (a, d, g & j), zone 2 (b, e, h & j) and zone 3 (c, f, I & l).

4.4 Discussion

Earlier in the thesis I discussed the direct effects of wildlife grazing on production of improved pastures. A number of the findings suggested that the indirect effect of wildlife grazing on pasture plant composition and production may be significant. As a consequence, a considerable amount of experimental research was undertaken to evaluate the influence of grazing on the botanical composition and ground cover of pastures, and to consider the implications for the long-term persistence and productivity of pastures.

Variability in the spatial distribution of grass species may be explained by a number of factors. Movements of fence lines since the initial sowing mean that the area in which the experiment was run has distinct pasture composition differences due to different management over the preceding years. It is unclear whether Zone 1 was ever sown with exotic species. However, it is now dominated by annual grasses including barley grass, silver grass and winter grass. If exotic grasses were sown, then grazing by wildlife and sheep using this area as a campsite may have contributed to the decline in desirable plant species. This zone can be characterised by overgrazing and further testing may reveal higher soil compaction than in other zones, as barley grass was in greater proportion than other zones. Kloot (1987) examined the influence of environmental factors on the germination and establishment of barley grass (*Hordeum glaucum*) and annual ryegrass (*Lolium rigidum*). Barley grass was more dominant than annual ryegrass due to the ability of barley grass to germinate on compacted ground and under high osmotic pressure (Kloot 1987)

Zone 2 was sown with a ryegrass and subterranean clover mix prior to 1970 (S. Foster pers. comm. 2010). However, this pasture appears to have degraded in species composition over the years, more so than in Zone 3. Wildlife grazing pressure in this zone appears to be intermediate between Zone 1 and Zone 3 and the persistence of ryegrass in low rainfall areas and throughout droughts would be low (Knox *et al.* 2006). Annual grasses may be better suited to survive droughts than perennials as they exist as seed, some dormant, which can germinate when soil moisture conditions improve in response to rain. This may have led to a change in botanical composition away from perennial grasses towards annual grasses and legumes. Studies by Robertson (1987) showed that annual grasses were encouraged by winter and moderate summer rainfall, while above average summer rainfall encouraged perennial grasses. Generally, low summer rainfall and high evaporation rates at Fosterville would suggest a similar favouring of annual grasses than perennials long term. Yates *et al.* (2000) found that livestock grazing negatively affected native perennial cover while exotic annuals increased in remnant grassy woodlands. Subterranean clover is advantaged by grazing during summer and autumn which reduces height of competing species and leads to increased germination of the subterranean clover seeds (Clark 1997). This may explain the greater cover of subterranean clover in this zone as the cover of desirable grasses was lower.

The pasture in Zone 3 was sown with a phalaris, cocksfoot and subterranean clover mix in the early 1980's (S. Foster pers. comm. 2010). However, like many pastures on the property, over time it has become dominated by phalaris. The ability of phalaris to survive dry summers and droughts, due to dormant underground buds (Lodge 1997), make it well adapted to persist in the low rainfall Midlands region. This zone appears less likely to be overgrazed as it is away from sheep camp sites and >550 m from the nearest edge of native vegetation edge. As a consequence it has a higher composition of perennial plants.

The observed variability in the spatial distribution of pasture species may have confused the effect of wildlife grazing on botanical composition, and means that zones cannot be compared with each other as they have different dominant pasture species. Ideally a site with only one sowing treatment would have been available for the study. Unfortunately no such trial site was present on this property and if it was established would not be representative of pastures in this region. Even in a site with only one sowing treatment, spatial variability is likely to occur over time due to variations in soil types and fertility, and sheep preferring to graze and camp in some areas of the paddock over others. Spatial variability in botanical composition by sowing treatment complicates the effect of distance on wildlife grazing as there may be preferences for some pasture species over others. This may alter the feeding behaviour of wildlife spatially.

Jones *et al.* (1995) discussed the limitations of experiments that monitor short-term changes in species composition, as some changes in species composition may take longer than 10 years. Shorter-term studies are likely to be influenced by seasonal changes and rainfall events (Robertson 1987).

Annual grasses such as barley grass (the main contributor in the current study) germinate in autumn and yield through late autumn, winter and early spring, senescing in late spring through to mid-summer, depending on rainfall. The palatability and digestibility of barley grass decline with onset of senescence, and seed heads can also contaminate wool (Knox 1999). The contribution of annual grasses during the winter and spring period increased relative to the perennial grass species and decreased significantly during summer. The seasonal contribution of other species in terms of % cover and DM in a pasture dominated by annual grasses will thus be affected by the seasonal fluctuations in annual grasses. For example, ryegrass DM contribution in enclosed plots in Zone 2 increased during summer 2010 even though total DM was decreased. This was due to the sudden decline in the contribution of annual grasses to DM as they senesced. The hand separation method therefore gives a more accurate estimation of the contribution of each pasture species at a given time. The responsiveness of cocksfoot to summer and autumn rains and the summer dormancy of phalaris can account for the small seasonal fluctuations of these species in Zone 3.

The continual increase in yield of cocksfoot in enclosed plots in Zone 2 throughout the trial period was examined further. The increase in yield suggested that plants were recovering from the possible effects of overgrazing by wildlife, and the observed steady increase in plant cover over the trial was consistent with this explanation. There did not appear to be a decline in the yield of cocksfoot at any stage, rather the yield reached a plateau during autumn 2009 and summer 2010. Heavy continuous grazing favours annual grasses by reducing competition from perennial grasses, as annuals can survive over summer as seed (Robertson 1987). If perennial grass species avoid grazing, particularly during summer, they may receive a competitive advantage over annual grasses.

The sustained yield of cocksfoot during the trial may also be attributed to plant cultivars. The cultivars of cocksfoot are thought to be a mix of ‘Currie’ and ‘Porto’ (S. Foster pers. comm.. 2007). While both may be considered intermediate types, Porto actively grows in winter and summer being an intermediate type between Mediterranean summer dormant and north-European non-dormant types (Register of Australian Herbage Plant Cultivars 1972b; Register of Australian Herbage Plant Cultivars 1972a). Coupled with above average summer and early autumn rains experienced at Fosterville in 2010 (see rainfall data in Chapter 2), the yield of cocksfoot would be expected to be higher compared with previous summers.

The large difference in yield of cocksfoot between enclosed and exposed treatments leads to the hypothesis that wildlife may preferentially graze cocksfoot. Cocksfoot may be the only pasture species actively growing and producing DM during the summer period as growth of ryegrass can be restricted by temperature or may enter dormancy (Knox *et al.* 2006), phalaris becomes dormant (McWilliam 1968; Register of Australian Herbage Plant Cultivars 1972c) and annual grasses senesce. Therefore it is likely that at a period when green herbage is scarce, cocksfoot would be under increasing grazing pressure from wildlife selecting green herbage. Heavy defoliation of cocksfoot depletes carbohydrate reserves and can limit regrowth in autumn (Volaire 1994). Grazing by wildlife may therefore restrict the yield of cocksfoot in a mixed pasture and this will be exacerbated in dry summers when green herbage is scarce. Heavy grazing just after growth recommences in autumn should be avoided (Virgona and Hill 1997), however, with uncontrolled wildlife this is difficult to manage. Over time, wildlife grazing would be expected to lead to a decline in the persistence of cocksfoot in the sward.

There are many examples reported in the literature of changes to the botanical composition of grasslands caused by preferential grazing. Heavy grazing by sheep has thought to have reduced *Themeda australis* in many rangelands in Australia (Robertson

1987). Studies by Neave and Tanton (1989) in the Tidbinbilla Nature Reserve in the Australian Capital Territory showed that preferential grazing by grey kangaroos on *Themeda australis* reduced the height and cover of this species. It was recommended following the study that management would be required to maintain the *Themeda* grassland either through kangaroo culls or fencing of the grassland (Neave and Tanton 1989). In the current study, grazing by wildlife limited the cover and DM production of cocksfoot available for sheep. As reported by Neave and Tanton (1989) for the grazing management of *Themeda*, increased management of the wildlife population at Fosterville will be required to allow the cocksfoot to recover within the pasture and ensure persistence. Sheep may also have a preference for cocksfoot within the pasture, however sheep can be removed before overgrazing occurs.

Fluctuations in subterranean clover cover between enclosed and exposed plots can be explained in two parts. Firstly, subterranean clover may have increased in enclosed plots in Zone 1 due to the removal of much of the canopy dominated by barley grass, as well as the protection offered by the enclosure, which would have ensured growth following late summer/early autumn rains. Secondly, higher levels of subterranean clover cover in Zone 2 in the exposed plots may indicate relatively less competition from grasses. Less canopy cover of grasses allows greater light penetration, which may aid in germination of the prostrate growing clover, which then proceeds through its annual life cycle before the process begins again in the next season. A similar mechanism may have been in operation in the study by Neave and Tanton (1989) where the cover of *Trifolium* sp. amongst other species, increased in response to a decreased height and cover of *Themeda australis*. This observation is consistent with the theory discussed by Caswell (1978) of predator-mediated coexistence - in this case grazing by wildlife has reduced the canopy cover of grasses allowing clover to co-exist or thrive in this system.

Studies by Leigh *et al* (1995) and Halsall *et al.* (1995) have shown the negative allelopathic effects of standing herbage and litter on the germination and root growth of subterranean clover. Leigh *et al* (1995) suggested that reducing plant residues prior to autumn rains may be essential for good germination of subterranean clover in mixed pastures. Thus the effects of shading due to greater pasture biomass and increased litter within exclusion cages may also have contributed to lower seed germination of subterranean clover. A long running study in New South Wales on the effects of stock grazing on a sown perennial pasture consisting of phalaris and white clover, showed significant changes in species composition after 30 years of varied stocking rates (Greenwood and Hutchinson 1998). Phalaris remained dominant in the ungrazed treatment while white clover was almost absent. In pasture under low stocking rates of sheep (10 animals/ha), phalaris was reduced to less than 40% of the basal cover, while more than 50% of the basal cover consisted of other grasses. However, the pasture under high stocking rates of sheep (20 animals/ha) was dominated by annual clovers (45%), other grasses (26%), and other dicotyledons (18%), while phalaris had declined to only 11% of the basal cover (Greenwood and Hutchinson 1998).

Fletcher (2006) recorded significant differences in ground cover between exclusion cages that had been in place 2 months and unprotected plots. On average there was 4% more ground cover under exclusion cages (Fletcher 2006). In the current study, there were large differences in the amount of bare ground in Zone 1. There were greater amounts of bare ground in exposed treatments than enclosed treatments, particularly in 2008. The effects of the drought during this time led to low herbage availability. Grazing pressure by wildlife would presumably have been significant. This led to the reduction in cover of annuals in exposed plots. Larger differences in the amount of ground cover may have been recorded by Fletcher (2006) had it been possible to have exclusion cages in place longer, as in the current experiment.

Grazing intensity also has an effect on ground cover. Schönbach *et al.* (2011) visually estimated soil coverage in Inner Mongolia and found that coverage decreased with increased grazing intensity. Schönbach *et al.* (2011) found that litter accumulation decreased with grazing intensity by as much as 83% for very heavy grazing intensity, compared with un-grazed plots. Similar studies by Oztas *et al.* (2003) in Turkey found that overgrazed sites had soil canopy coverage percentage of over 30% which was susceptible to water erosion. This clearly shows the issues associated with un-controlled wildlife grazing on pastures close to vegetation, especially when exacerbated by grazing demands from sheep in times of drought. With less cover, susceptibility to erosion from heavy rains increases at least in the short term following drought. The lack of perennial species in Zone 1 may render this zones susceptible to erosion from heavy rains during the summer months. Retaining some standing dead matter and litter may be important in preventing such events. Neave and Tanton (1989) collected data on the amount of bare ground in plots grazed by kangaroos, but did not present this in their results. However, the reduction in cover (%) of *Themeda* in the sward appeared to result in an influx of other grassland species rather than an increase in bare ground. This response may have been related to the higher overall species richness in the grassland in that study compared with the current study where few species were present. A limited seed bank reserve in my study may have contributed to the low level of re-colonisation by pasture species that was observed.

Differences between cocksfoot in enclosed and exposed plots in Zone 2 and Zone 3 may be explained by plots originally having more cocksfoot plants present in the enclosed plots. While cages were randomised it is possible that the small replication size was affected by variability in original species composition. As the trial started in February 2008 and the first harvest was in June 2008 it is difficult to tell whether protecting the pasture during that time caused differences in the composition of cocksfoot and other species prior to the first harvest. This period in 2008 was during the drought and herbage availability was low. Pressure on high nutritional species such as cocksfoot, prior to and during this period, would have been significant from both sheep and wildlife due to below average rainfalls in 2006 and 2007 (see Table 2.2). Therefore, it could be hypothesised that heavy grazing of cocksfoot by wildlife during this time may have resulted in a lag in cocksfoot yield during the subsequent seasons (2008 and 2009).

Investigation of perennial grasses for comparative ability to withstand overgrazing by wildlife while maintaining ground cover and DM production for sheep is warranted. Investigation of the effects of overgrazing by wildlife on the below ground biomass of roots may indicate the pasture's ability to recover from overgrazing. Furthermore, the effects of the reduction in ground cover and above ground biomass on soil health may be important for pasture persistence and these effects are examined in Chapter 5.

4.5 Conclusions

The first aim was to quantify the effects of wildlife grazing on pasture species composition and ground cover in pastures. I quantified the effects of wildlife grazing on pasture species composition and demonstrated that grazing has both direct and indirect effects on both the growth and likely survival of pasture species. The difference in observed and modelled pasture growth rates in enclosed plots indicated that there may have been a number of indirect effects of wildlife grazing on the botanical composition of the pastures and the extent of ground cover. The composition of pastures by dry weight and percentage cover varied seasonally. Wildlife grazing can alter the composition of pastures by reducing the cover and yield of improved grasses. Bare ground was most evident near the edge of native vegetation where grazing was highest.

The second aim of this chapter was to test for correlations between wildlife grazing and a decline in the composition of improved pasture species over time, and the third (and related) aim was to test if wildlife grazing may contribute to an increase in the cover of less desirable grasses and broadleaf weeds. My research indicated that wildlife grazing may lead to the death of individual pasture plants and increase the area of bare ground present in pastures. These changes can provide an opportunity for less desirable grasses (for production purposes) and broadleaf weeds to colonise.

The final aim of this chapter was to quantitatively determine if protecting pastures from wildlife grazing can increase ground cover and reduce the susceptibility of pastures to erosion. My research indicated that controlling wildlife during summer may be particularly important in preventing summer-active grasses such as cocksfoot from being overgrazed. It would appear that the effects of wildlife grazing can be exacerbated during herbage shortages, especially during drought. Overgrazing by wildlife in combination with routine sheep grazing can reduce ground cover and increase susceptibility to erosion. Selective grazing by wildlife in addition to grazing by sheep, particularly during drought, can lead to degradation of the pasture, losing improved perennial species first. Reductions in canopy cover of improved and annual grasses may result in an increase in the cover of subterranean clover.

Chapter 5: Influence of wildlife grazing on soil health in an established perennial pasture

5.1 Introduction

Soils provide the structure, fertility, and moisture that pasture plant species require to grow. Therefore maintaining good soil health is necessary for pasture productivity. Definitions of soil health refer to a soil's capacity to sustain productivity while maintaining plant, animal and human health (Arias *et al.* 2005; Cotching 2009b). Soil health was defined by Doran and Safley (1997) as: 'the continued capacity of a soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain biological productivity, promote the quality of air and water environments and maintain plant, animal and human health'. The term 'soil health' has sometimes been preferred over soil quality as it is a better descriptor of the dynamic nature of soils as a living organism (Doran and Safley 1997).

In order to assess soil health, indicators of soil condition are measured. Physical indicators include depth of soil, texture, soil bulk density, water holding capacity, and rooting depth of plant species. Measurable chemical attributes include soil organic matter, electrical conductivity, and extractable macro- nutrients: nitrogen (N), phosphorus (P), and potassium (K). Biological indicators of soil health relate to the content and processes of soil microbes. Measuring bacterial and fungal biomass, soil respiration and temperature are also regarded as good indicators of soil health (Doran and Safley 1997).

These indicators can be used to develop a current status of a soil's health (Arias *et al.* 2005). Indicators can be compared with standards or normal ranges for each attribute. In the long-term, repeated tests of soil health indicators provide the data necessary to monitor soil health in response to different management practices. Landowners can then adjust their management accordingly. One of the purposes of measuring soil health is to protect and improve long-term agricultural productivity (Cotching 2009b).

Microbial biomass has been described as being the living component of soils excluding plant roots and macrofauna (Jenkinson and Ladd 1981; Sparling 1997). It includes a wide range of organisms such as fungi, bacteria, protozoa and nematodes. Microorganisms are critical in the process of organic matter breakdown and release of nutrients, and as such can influence soil structure and nutrient availability (Sparling 1997). Monitoring the levels of microorganism in soils can provide a good indication to changes in soil health (Sparling 1997).

Studies of microbial indicators on pastures in Tasmania have shown fungal/bacterial ratios of 1.6 in the north-west and 1.9 in the northern Midlands (Cotching 2009a). McDonald *et al.* (2010) found that the fungal/bacterial ratio averaged 1.25 in soils on a range of Tasmanian farms, and was generally higher in autumn than in spring. The studies of Cotching (2009a) and McDonald *et al.* (2010) indicated that microbial activity can vary depending on soil moisture, temperature, and organic matter and that comparison of biomass levels and ratios must be made under similar conditions or times of year. Furthermore, the study by McDonald *et al.* (2010) reported that while fungal biomass increased from autumn through to spring, bacterial biomass remained constant under dryland pasture. In contrast, Cotching (2009a) found that levels of bacterial biomass increased significantly between November and April, while fungal levels remained steady.

Soil health is strongly influenced by management. Management factors that have been shown to influence soil health parameters include grazing intensity (Yates *et al.* 2000; Drewry and Paton 2005), applications of fertilisers (Bünemann and McNeill 2004), applications of pesticides (Van Zwieten 2004), and tillage practices (Balota *et al.* 2003).

Grazing management can influence the characteristics of the soil. Grazing can result in a decline in root biomass (Donaghy and Fulkerson 1998), thus multiple grazings in quick succession are likely to have an impact on root biomass. High stocking rates or overgrazing can lead to soil compaction particularly when soils are wet. Yates *et al.* (2000) found that increased grazing intensity in woodlands increased soil bulk density and soil penetration resistance. In addition Drewry *et al.* (2005) found that trampling of wet soils by sheep at high stocking rates reduced macroporosity of soils in newly sown pastures, however it did not result in a decline in pasture growth. Soil becomes compacted by increasing the density of soil particles and reducing the amount of soil pores (spaces between particles). Soil pores are important for soil health as they enable the flow of water, air and the growth of plant roots through the soil. Soil compaction can thus lead to a decline in pasture productivity as plant roots have restricted access to water and nutrients.

Previous studies described in this thesis have shown that grazing wildlife have a significant effect on the above ground production of biomass (Chapter 3) and can influence plant species composition of pastures. The intention of the current study was to determine if wildlife grazing can influence soil health, by measuring a number of soil properties and indicators.

The specific aims of this chapter were to:

- Test for correlations between wildlife grazing and soil health and root biomass in pastures;
- Quantify the effects of wildlife grazing on soil health indicators such as levels of soil carbon, soil nitrogen, pH and soil electrical conductivity;
- Quantify the effects of wildlife grazing on the microbial biomass within soils; and
- Test for correlations between wildlife grazing and the root biomass of grazed plants.

5.2 Methods

A full description of the experimental field site and experimental field design was given in Chapter 3.

5.2.1 Soil sample collection

Soil nutrient testing

Soil health and soil fertility were assessed at the completion of the pasture biomass harvests in April 2010. All harvested plots were soil sampled using a 100 mm ‘pogostick’ soil corer (Figure 5.1f) and a 500 g sample was dried and sent to CSBP Soil and Plant Analysis Laboratory for analysis of nitrate N (NO_3^-), ammonium N (NH_4^+), total N, organic carbon (C), total C, pH (H_2O and $CaCl_2$), and electrical conductivity. Samples were oven-dried at 40°C for 2 days. A summary of the methods used by CSBP follows:

Soil nitrate nitrogen and ammonium nitrogen were extracted with a 1M potassium chloride solution for 1 hour at 25°C. After dilution the resulting soil solution was measured on a Lachat Flow Injection Analyser. The concentration of ammonium nitrogen was measured colorimetrically at 420nm using the indo-phenol blue reaction. Nitrate was reduced to nitrite through a copperised-cadmium column and the nitrite was also measured colorimetrically at 520 nm (CSBP 2010).

The Walkley Black method was used to determine soil organic carbon content. Concentrated sulphuric acid was added to soil wetted with dichromate solution. The heat of the acid-based reaction was used to induce oxidation of soil organic matter (OM). Chromic ions produced were proportional to oxidized OC and were measured colorimetrically at 600nm on a Multiscan (CSBP 2010).

Soils are extracted in deionised water for 1 hour to achieve a soil solution ratio of 1:5. The water pH and electrical conductivity of the extract was subsequently measured using a combination pH electrode. After water pH and electrical conductivity have been measured, calcium chloride solution was added to the soil solution and after thorough mixing the calcium chloride pH was determined (CSBP 2010).

Microbial biomass

A soil sample was bulked for each treatment at each boundary. Samples were collected using a 100 mm ‘pogostick’ soil corer (Figure 5.1), stored in a refrigerator and sent to

CSBP soil analysis laboratory in an overnight shipment to ensure integrity of the sample. Samples were sent for boundaries 1, 3, 5, 7 and 9 for both the exposed and caged treatments. Samples from boundaries 2, 4, 6, and 8 were not sent due to the cost of the analyses. Microbial tests were carried out by The ERA Sustainable Farming Co. (ERA 2010a; ERA 2010b).

Samples were first sieved (1 mm) and incubated at 25°C for 36 hours prior to treatment. Fresh soil equivalents to 2 g dry weight were weighed into specialised vials and treated with one of the following: fungal inhibitor only, bacterial inhibitor only, mixture of fungal and bacterial inhibitors, or control with no inhibitors. The treatment mixtures were thoroughly mixed with soil. Glucose was added to each vial after 1 hour of incubation at 25°C, mixed thoroughly, stoppered with rubber bungs and re-incubated. After 4 hours of incubation, CO_2 gas accumulated in each vial was collected and injected in an infra-red gas analyser. Prior to measurements, the analyser was calibrated with 5% ultra pure standard CO_2 gas. The biomass was calculated by measuring peak heights and applying the following equation:

μg CO_2-C g-1 soil = ((Asample x 10000/Bstd)/103 x V x K)-((Ablank x 10000/Bstd)/103 x V x K)

Where: Asample = Peak height (mm) of sample

Ablank = Peak height (mm) of blank bottle

Bstd = Peak height (mm) of standard gas (1% CO_2)

V = Head space volume of bottle

K = Conversion constant for μl to μg of CO_2-C (assuming 1 atmosphere pressure: 1.7995 μg

CO_2 = 1 μl CO_2 at 25 °C and 1 atmosphere = 0.4908 μg CO_2-C)

Fungal biomass, bacterial biomass, fungal-bacterial ratio and inhibitor additivity were calculated based on the respiration rate in the presence and absence of microbial inhibitors (ERA 2010b).

Figure 5.1 The root biomass core collection process; a) Bruce Dolbey driving in the 40 mm diameter tapered corer, b) extracting the corer, c) removing the core from the corer, d) Rowan Smith sectioning the core into upper 150 mm and lower 150-300 mm and cutting off excess length, e) soil core with upper and lower 150 mm sections, f) collecting basic soil test cores with a 100 mm 'pogo stick' soil corer.

Root biomass

A 40 mm diameter soil corer was used to obtain two 300 mm length cores per plot which were subsequently split into 0-150 mm and 150-300 mm length cores for assessment. The soil corer was systematically placed within the quadrat (Figure 5.1a). If a rock impeded coring then another random position was used.

Soil cores were dried for 2 weeks at 40 °C in a drying room. Cores were crushed using a mortar into fine aggregates and sieved through a 4 mm sieve. Samples were then weighed and placed in a tray with 1L of warm water with detergent for 20-30 minutes. Roots were then separated by a process of sieving (1 mm and 0.5 mm sieves) and skimming the floating roots from the soil and water (Figure 5.2). Organic matter was removed from the roots with a pair of forceps. This method was adapted from methods used by Jackson and Bloom (1990) and Kücke *et al.* (1995) for extracting roots from soil. Up to 90% by weight of roots can be recovered by using a floatation and sieving method (Pearson and Jacobs 1985). Roots were dried at 40 °C for 24 hours to remove water. Roots were then dried at 60 °C for 48 hours and weighed for DM yield.

Figure 5.2 The root washing and sieving process; a) washing the soil sample through 1000 µm and 500 µm sieves to remove silt particles, b) using plastic tub to collect waste water and soil, c) pouring the floating roots off the remaining slurry into the 500 µm sieve, d) washing the roots from the sieve into a collection jar, e) collecting the remaining roots with forceps, f) roots dried on trays at 40 °C for 24 hours then 60 °C for 48 hours.

5.2.2 Data analysis

A linear mixed model (PROC MIXED) was used to analyse soil nutrient and root biomass data with distance from the bush edge. The model was fitted to the data with distance (boundary) and treatment (enclosed or exposed) and distance by treatment terms treated as fixed effects. Distance by paired plots was the random term used to test the distance effect, while the other fixed effects were tested with the residual error.

A Paired-Samples T-Test was performed using statistical package SPSS (Version 18, SPSS Corporation, Illinois, USA) to separate the differences in microbial biomass levels between enclosed and exposed treatments.

5.3 Results

5.3.1 Soil nutrient status

An analysis of variance showed there was no significant difference between the concentration of soil nutrients, for example ammonium nitrogen ($F_{1,54}$ = 2.809; P>0.05), as well as electrical conductivity and soil pH from soils that were protected from wildlife grazing compared to those exposed to grazing (Table 5.1). When enclosed and exposed values were then pooled, mean (± SEM) soil ammonium N was 26.15 (± 2.98) mg/kg, soil nitrate N 22.57 (± 3.08) mg/kg, total N 0.35(± 0.02) %, organic C 3.43 (± 0.16) %, conductivity 0.11 (± 0.01) dS/m, and pH (H_2O) 5.5 (± 0.06).

Data within each distance boundary (both enclosed and exposed plots) were pooled to examine the effect of distance from the edge of native vegetation on soil nutrients levels. The level of nitrate N decreased with distance from native vegetation and is displayed in Figure 5.3(a). Nitrate levels were highest (41.13 ± 7.22 mg/kg) in the 25 m enclosed treatment and lowest (10.50 ± 1.28 mg/kg) in the 800 m enclosed and exposed treatments. There was no identifiable trend between ammonium N levels and distance from native vegetation (Figure 5.3b). Figure 5.3c and Figure 5.3d show that total N and organic C percentages decreased with distance from native vegetation edge. However, electrical conductivity was highest close to the edge of native vegetation (0.15 ±0.01 dS/m at 25 m) and decreased to 0.08 ±0.01 dS/m (250 m) (Figure 5.3e). Soil pH levels were relatively consistent ranging from 5.2 ±0.04 (250 m) to 5.74 ± 0.09 (650 m) (Figure 5.3f).

Table 5.1 The significance of fixed effects of distance, treatment and distance by treatment on soil nutrients.

	Distance		Treatment		Distance*Treatment	
	F value	P value	F value	P value	F value	P value
Ammonium nitrogen	3.115	0.012	2.809	0.105	0.415	0.902
Nitrate nitrogen	4.615	0.001	0.286	0.597	0.675	0.709
Total nitrogen	8.076	<0.001	0.009	0.924	0.497	0.847
Organic carbon	7.183	<0.001	0.066	0.799	0.573	0.791
Conductivity	6.727	<0.001	0.112	0.741	0.357	0.943
pH $CaCl_2$	9.170	<0.001	0.478	0.495	0.951	0.493
pH H_2O	7.157	<0.001	0.207	0.653	1.138	0.371

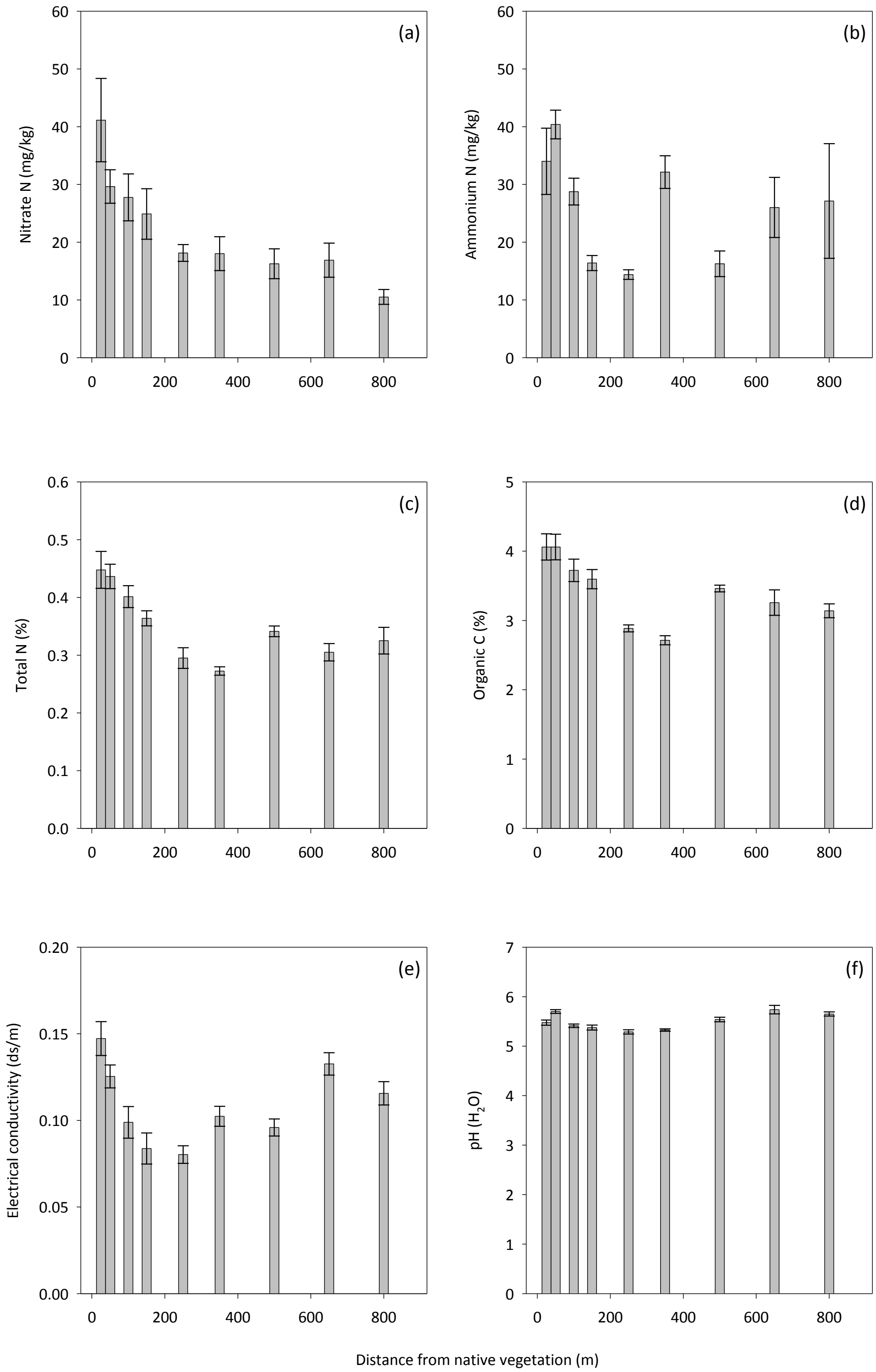


Figure 5.3 Relationship between distance from native vegetation edge and soil chemical properties; a) nitrate nitrogen, b) ammonium nitrogen, c) total nitrogen, d) organic carbon, e) electrical conductivity, and f) soil pH. Error bars represent the standard error of the mean.

5.3.2 Microbial analysis

Analysis using a paired-samples t-test indicated that there was no significant difference between the measured microbial parameters from soils that were protected from wildlife grazing compared to those exposed to wildlife grazing (Table 5.2). For example, there was no significant difference ($t_4 = 0.688$; P = 5.29) in bacterial biomass, or significant difference ($t_4 = 1.900$; P = 0.130) in the fungal/bacterial ratio between exposed and enclosed plots. The difference in total active microbial biomass (TAMB) between enclosed (34.9 ± 4.76 ug C/g) and exposed (25.0 ± 1.50 ug C/g) plots was close to significant ($t_4 = 0.925$; P = 0.063).

Table 5.2 Soil microbial bioassay results for exclosure and exposed plots. Means (± SEM) of biomass variables of 5 distance boundaries; 25 m, 100 m, 250 m, 500 m, and 800 m. df = 4. A paired t-test was used to compare means.

	Mean		t Value	P Value
	Enclosed	Exposed		
Bacterial biomass (ug C/g)	12.6 (±5.00)	9.1 (±1.15)	0.688	0.529
Fungal biomass (ug C/g)	13.2 (±3.46)	8.4 (±2.45)	1.133	0.320
Unknown biomass (ug C/g)	9.1 (±4.17)	7.5 (±3.57)	0.271	0.800
Total active microbial biomass (ug C/g)	34.9 (±4.76)	25.0 (±1.50)	2.548	0.063
Fungal/Bacterial ratio	1.4 (±0.20)	0.9 (±0.21)	1.900	0.130
Moisture content (%)	22.1 (±1.95)	20.5 (±2.17)	0.925	0.407

5.3.3 Root biomass

An analysis of variance showed that there was no significant difference between root biomass in the 0-150 mm ($F_{1, 25} = 0.000$; P = 0.992) and 150-300 mm ($F_{1,15} = 0.295$; P = 0.595) root zones of enclosed and exposed plots (Table 5.3). However, there was a significant distance from native vegetation effect on the root biomass at 0-150 mm ($F_{8, 27} = 13.848$; P<0.001) and 150-300 mm ($F_{8, 27} = 10.940$; P<0.001). There was no significant ($F_{8, 25} = 1.685$; P = 0.152) ($F_{4, 15} = 2.344$; P = 0.102) interaction between enclosed and exposed plot treatment and distance to native vegetation for root biomass at both 0-150 mm and 150-300 mm root zones respectively (Table 5.3).

Table 5.3 Analysis of root biomass data for fixed effects; distance (0-150 mm F = 8, 26-27; 150-300 mm F = 4, 15), treatment (0-150 mm F = 1,25)(150-300 mm F = 1,15) and boundary*treatment (0-150 mm F = 8,25)(150-300 mm F = 4,15)

	Distance		Treatment		Distance*Treatment	
	F value	P value	F value	P value	F value	P value
0-150 mm	13.848	<0.001	0.000	0.992	1.685	0.152
150-300 mm	10.940	<0.001	0.295	0.595	2.344	0.102

The root biomass in the 0-150 mm root zone increased significantly ($F_{8, 27}$ = 13.848; P<0.01) with distance from native vegetation (Figure 5.4). The combined treatment mean root biomass was 5.90 t DM/ha at 100 m and increased to 16.18 t DM/ha at 800 m. Root biomass was lower at the 25 m and 50 m distance boundaries but cores were only sampled to 100 mm due to the shallow depth of the soil. Soil cores were not attainable to a depth of 300 mm due to the presence of rocks in the 25, 50, 100, and 150 m distance boundaries. Root biomass was significantly (P<0.05) lower in the 150-300 mm root zone compared with the 0-150 mm root zone for all distances sampled (Figure 5.4). Similar to the 0-150 mm results, root biomass at 150-300 mm increased significantly ($F_{8, 27}$ = 10.940; P<0.01) between the 250 m and 800 m distance boundaries.

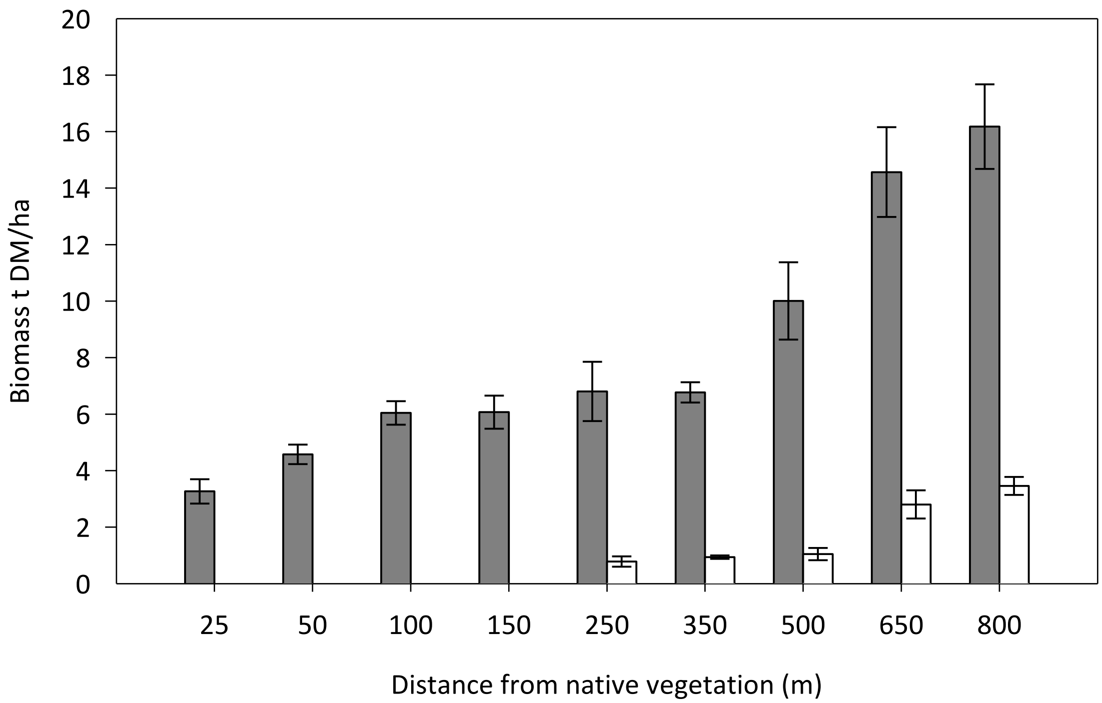


Figure 5.4 Relationship between distance and root biomass of depths 0-150 mm (■) and 150-30 mm (□)(enclosed and exposed plots combined). Note: cores were sampled to 100 mm in 0-150 mm depth at distances 25 m and 50 m and no cores were sampled at 25 m, 50 m, 100 m, and 150 m in 150-300 mm due to the shallow soil depth and presence of rocks. Error bars represent the standard error of the mean.

5.4 Discussion

It was hypothesised that grazing by wildlife may influence the soil health of the plots exposed to grazing. Soil chemistry and soil biology are processes which can contribute greatly to the productivity of pastures. It was also hypothesised that grazed plots which had less pasture biomass and less ground cover would have lower nutrient levels, less microbial activity and less root structure than plots protected by exclusion cages.

The soil analyses did not reveal a relationship between levels of ammonium N, nitrate N, total N, organic C, pH, or electrical conductivity and grazing treatment. The plot areas were 0.5 m x 0.5 m which may have been too small to pick up differences in nutrient status of enclosed and exposed plots. Mobile nutrients may readily move through the soils, particularly in the plots close to the native vegetation edge where there was a topographical gradient. Higher levels of N in the soil close to the native vegetation may be explained by sheep using these areas as camp sites and thus having a greater concentration of urine and dung deposits (Landsberg and Wylie 1988).

The microbial biomass within a soil is a key component in processes such as the breakdown of organic matter and the mineralisation of nutrients into plant available forms (ERA 2010a). The amount of organic matter in a soil is influenced by climate, soil type, topography, tillage and importantly vegetative growth (Cotching 2009a). Greater production of plants results in higher levels of organic matter in the soil. Pastures contribute organic matter over a long period of time and this allows an increase in organic matter to higher levels than in fields left fallow.

It was hypothesised that as production was greater in the enclosed treatment plots (Chapter 3) then both greater levels of organic C and total microbial biomass would be expected. The analysis of TAMB showed that higher levels in the enclosed than the exposed treatment were close to significant ($P = 0.063$). The lack of a stronger significant ($P<0.05$) difference between enclosed and exposed plots could be due to the improved growing conditions over the last 9 months of the experiment. Above average rainfall in June, July, August, and September (See Table 2.1) led to an increase in DM yield. This increased available feed for wildlife in the woodland areas of the Macquarie Tier and is likely to have reduced the grazing pressure on the experimental area. With increased pasture production occurring on exposed plots the differences between treatments may have declined. This could explain why the fungal to bacterial ratio, fungal biomass, bacterial biomass and other fauna were all higher in the enclosed treatment than the exposed treatment, but no significant difference was detected at the time of sampling.

Total active microbial biomass levels between 10-45 ug C/g are considered to be adequate for good soil health (ERA 2010a). Levels recorded for enclosed and exposed plots fit within the 'high' range of 25-35 ug C/g for winter (ERA 2010a). Such high levels recorded here would suggest that these processes are functioning efficiently. The fungal:bacterial ratio results fit within the range 0.5-1.5, a level considered by ERA (2010a) to be ideal for organic matter breakdown and nutrient cycling.

The level of microbial biomass within the soil generally reflects the level of organic matter within the soil (Sparling 1997). Enclosed plots yielded more DM and thus were hypothesised to contribute greater amounts of organic matter to the soil and have higher levels of microbial biomass. While only a weak relationship was suggested between treatments and TAMB levels, an experiment detailing the changes in soil health over a longer time frame, with more intensive sampling and in larger treatment plots may have produced significant differences.

There was a distinct increase in root biomass in the 0-150 mm root zone with distance from native vegetation. The recorded variation in root biomass may be explained by the dominant pasture species found within each of the zones identified in Chapter 4. The highest root biomass was found in the area dominated by phalaris which is a deep-rooting and drought tolerant species. Lower levels of root biomass are found in zones dominated by annual grasses or perennial ryegrass which are shallower rooted species (Blair 1997). Consistent with similar studies (Mapfumo *et al.* 2002; Lodge and Murphy 2006), the majority of the root mass was found in the 0-150 mm root zone. Greenwood *et al* (1998) found that greater proportions of roots were found closer to the soil surface under higher grazing intensity. However, these may have been related to botanical composition of treatments as botanical composition of pastures changes under different stocking rates. Although botanical composition changes were recorded (Chapter 4), botanical composition changes in pastures generally occur over many years. Evans (1973) found that repeated defoliations of pasture grasses to 25 mm caused root elongation to either cease or nearly cease. More immediate changes in root biomass following defoliation may be explained by the process of root regrowth having a lower priority for energy allocation than leaf regrowth for example (Donaghy and Fulkerson 1998).

My initial hypothesis was that protecting pastures from grazing would have a positive effect on root structure and biomass. However, protecting pastures had no effect on root biomass over the two years of grazing treatment. It is possible that 2 years may not have been long enough for differences in root weight to be expressed. In a study on grazing intensity treatments in New South Wales, Lodge *et al.* (2006) found that after 4 years of grazing there were few significant differences in root mass and other root related parameters. In contrast, a 10 year imposed grazing intensity study into the effects of grazing on mesofauna in pastures showed significant differences in the amounts of root biomass when comparing grazing intensities of 10, 20 and 30 sheep/ha (King and Hutchinson 1976). These may also have been related to botanical compositional changes in the pasture. The sampling method may also have contributed to the lack of significant differences. Randomly sampling only two cores per plot, potential differences in plant species sampled, and the high chance of hitting bare ground are possible explanations. This could have been overcome by taking cores of individual plant species and comparing between enclosed and exposed treatments.

5.5 Conclusions

This chapter had four aims. The first aim was to test for correlations between wildlife grazing and soil health and root biomass in pastures, and the second (and related) aim was to quantify the effects of wildlife grazing on soil health indicators such as levels of soil carbon, soil nitrogen, pH and soil electrical conductivity. Based on the analyses of my research, no correlations between wildlife grazing and soil health were identified, and wildlife grazing had no apparent major influence on the health of soils examined during the two year study period. The third aim of the chapter was to quantify the effects of wildlife grazing on the microbial biomass within soils. My research suggested that protecting the pasture from grazing had no (measurable) effect on soil C, soil N, electrical conductivity or soil pH. The fourth aim of the chapter was to test for correlations between wildlife grazing and the root biomass of grazed plants. My research suggested that wildlife grazing had no effect on microbial and fungal biomass levels and fungal/bacterial ratio, but had a weak effect on the total active microbial biomass.

Overall, it is possible that the duration of my study was not sufficient to establish significant changes to soil health by protecting pasture from grazing. Moreover, it is possible that any potential correlations between wildlife grazing and soil health were ‘masked’ or unable to be detected due to the fact that the pasture growing conditions found at Fosterville over the last 9 months of the study improved due to above average rainfall. I have suggested that the relationships between wildlife grazing and soil health require further investigation, and that longer term studies will be required for this purpose (see Chapter 8).

Chapter 6: Effects of grazing by wildlife on the establishment of improved pastures

6.1 Introduction

The establishment of new pastures and the renovation of run down pastures are essential for the sustainable production of grazing systems. Pasture productivity is generally increased by improving the botanical composition of desirable plant species, reducing undesirable weed species, and improvements in soil fertility. Successfully establishing new pastures is important because it influences the production potential of the pasture for further seasons. Knox *et al.* (2006) suggested that a perennial pasture should remain productive for at least 20 years, indicating that in the order of 5% (by area) of the property could be improved or re-established each year. However, variation in weather and other factors may prevent the establishment of pastures in some years while in favourable years larger tracts of pasture may be re-established.

The establishment of new pastures has the potential to improve farm production. Scott (1997) suggested that animal production could be increased by 250% given the use of suitable pasture plant species, favourable rainfall and nutrient levels, when compared with unimproved native pasture. Once established, grazing management of this improved resource is likely to influence profitability. However, it is not uncommon for much smaller increases in production to be reported in the literature. For example, a study in south-western Victoria, Australia, showed that re-sowing with improved pasture species phalaris, ryegrass, and subterranean clover and increased fertiliser rates resulted in an 18% increase in pasture production when compared with fertilised annual-based pastures (Saul *et al.* 2009). Saul *et al* (2009) reported that the proportion of sown perennial species rapidly declined over six years from 30 to 10% and suggested that set stocking could have contributed to this decline. Such a result would indicate the importance of grazing management in ensuring the persistence of a pasture.

Pasture establishment can be defined as all stages of seedling development from seed imbibition and germination through to seedling development and the stabilisation and survival of seedlings (Bellotti and Blair 1985). Campbell *et al.* (1987) identified key stages in the establishment of pastures. Each stage involves a number of factors that could influence the success of pasture establishment. These phases are: pre-sowing, germination, radicle entry, emergence, seedling growth and survival, and young plant growth and survival. Factors that influence the success of pasture establishment have been described by Campbell *et al.* (1987) and Scott (1997) and include soil moisture, temperature, and soil fertility. However, of particular interest is the effect of pests on the development of seedlings and survival of young plants.

Minimising the effects of pest damage, whether it is invertebrate or vertebrate, during establishment is viewed as being critical to achieving a persistent productive sward. Pasture DM loss caused by insects can vary between 0% and 88% (Allen 1987). In Tasmania, invertebrate pests such as corbie (*Oncopera intricate*), cockchafer (*Acrossidius* sp.) or (*Adoryphorus couloni*), red-legged earth mite (*Halotydeus destructor*), and wingless grasshopper (*Phaulacridium vittatum*) are controlled mainly through the use of registered insecticides (McQuillan *et al.* 2007).

Several published studies on the feeding behaviour and diet of macropods indicate that pastures can provide an important food source and that animals may spend significant periods of time feeding on pastures when they are located near vegetation suitable for shelter. Jarman and Phillips (1989) studied a community of macropods in the New England tablelands of New South Wales. They reported that eastern grey kangaroos appeared to prefer pasture and open forest while red-necked wallabies preferred wet and dry sclerophyll forest and adjacent pasture. Monocots made up 99% and 84% of the diet of the eastern grey kangaroo and red-necked wallaby, respectively. The dietary overlap of the two species was estimated to be 73.9% (Jarman and Phillips 1989). These authors also indicated that eastern grey kangaroos may preferentially feed on swards of short, green leafy grasses and that most macropods select for leaf in preference to sheath or stem when eating grass. Davis *et al.* (2008) examined the foraging behaviour of the eastern grey kangaroo at Wilsons Promontory National Park, Victoria. They also reported that the diet of the species consisted mainly of monocots and animals tended to consume leaf material compared with stems. The apparent preference of leaf material is most likely due to the DM digestibility of young grass is much greater than mature grass (Dawson 1989). Clearly, the establishment of new pastures next to remnant dry sclerophyll forest and grassy woodlands known to support macropods will increase the food resources of these animals and attract them to feed.

The influence of wildlife grazing and browsing on vegetation in Tasmania has included reduction of tall herbs (Bridle and Kirkpatrick 1999), reduction in flower heads (Bridle and Kirkpatrick 2001), and change to tussock grasslands (Scott and Kirkpatrick 2008). Other studies have focused on wildlife browsing in plantations (Newsome 1971; Bulinski and McArthur 2000b; le Mar and McArthur 2005; While and McArthur 2005; While and McArthur 2006) and trees on farms (Statham 1992). Studies by Statham and Raynor (1995), Donaghy and Tegg (2001) and later Smith (Chapter 3 and Smith *et al.* (2012)) presented in appendix 4 have attempted to quantify the effects of wildlife grazing on improved pastures in Tasmania, each showing varied production losses. These studies have focussed on the effects of browsing on already established pastures. However, anecdotal evidence suggests that establishing pastures are under increasing pressure by wildlife leaving the profitability of improving pastures in jeopardy. Many land owners are now apprehensive about undertaking pasture renovation in areas known to support large number of browsing wildlife. Although this is viewed as a risk mitigation strategy, degraded pastures are often seen to be limiting farm productivity and as such efforts are required to quantify the success of pasture renovations in the presence or absence of browsing pests. Efforts are also required to identify pasture species that are less palatable to wildlife or are able to withstand continuous grazing by wildlife. Such pasture species would be particularly useful to graziers in areas where there is severe browsing damage.

Production losses on establishing pastures are predicted to be greater than recorded in established pastures due to the greater vulnerability of the younger plants as well as lower pasture biomass leading to higher wildlife grazing pressure. To determine the influence of vertebrate browsing on establishing pastures, an experimental study was duplicated at two sites. One experiment was undertaken at Fosterville and the second within a wallaby enclosure at the DPIPWE's Mt. Pleasant laboratories site in Launceston, Tasmania. It was hypothesised that during the establishment of a pasture sward the wildlife grazing population, the pasture species sown, and the environment would have a significant effect on productivity and botanical composition of the establishing sward.

This chapter, along with the following chapter quantitatively examine establishing pastures and investigate the impacts of wildlife grazing on pasture establishment and options for mitigating grazing impacts through the implementation of wildlife control measures. I also investigated the influence exclusion cages have on pasture growth. The specific aims of this chapter were to:

- Quantify the effects of wildlife grazing on the accumulated biomass of four pastures types used for establishing pastures in the study region;
- Test for correlation between wildlife grazing and the botanical composition of improved pasture species, and the amount of bare ground within a pasture;
- Quantify the effects of exclusion cages on accumulated biomass of pastures.

6.2 Methods

Two sites were chosen to evaluate the impacts of wildlife grazing on pasture establishment with the two sites varying in climate, surrounding vegetation and wildlife composition. The experimental study was undertaken in two distinct stages. Stage one examined the effects of wildlife grazing on pastures during establishment. Stage two examined the effect of discontinuing wildlife control and introducing wildlife control one year after sowing. This second stage is reported in Chapter 7.

In addition, an investigation into the effects of exclusion cages on pasture growth was undertaken. This work is reported below.

6.2.1 Experimental sites

Fosterville site

The site was established within a degraded pasture and 100 m from the pasture - native vegetation (Macquarie Tier) interface at Fosterville (Figure 2.2). Daytime and spotlight observations, faecal pellets, tracks and infrared digital scouting cameras (ScoutGaurd SG550/SG530; HCO Outdoor Products, Norcross, GA) were employed to identify species grazing/browsing on the experimental site. Although difficult to estimate absolute abundance, an indication of wildlife numbers was recorded. As many as 20 wallaby were observed grazing at night on the experimental site. Mobs of up to 10 kangaroos were also observed in late afternoon and deer were also spotted in twos and threes. There were also faecal pellets found of rabbit and possum. It is likely that the number of animals grazing within the experimental areas varied during the study due to pasture availability and digestibility.

The existing pasture was surveyed in July 2008 by visual quadrat estimates of species presence and percentage cover. The existing pasture was dominated by annual grasses such as barley grass and winter grass as well as broadleaf weeds including erodium (*Erodium moschatum*), capeweed (*Arctotheca calendula*), and a number of thistles including slender (*Carduus* spp.), variegated (*Silybum marianum*), and saffron (*Carthamus lanatus*) species.

A detailed description of the upper soil profile is provided in Table 6.1. The soil fertility of the experimental site prior to commencing was as follows: Olsen phosphorus (P) 13.9 mg/kg, Colwell potassium (K) 288 mg/kg, mono-calcium phosphate extracted sulphur (S) 13.4 mg/kg, pH (H_20) 5.4 and electrical conductivity 0.191 dS/m.

Table 6.1 Soil description of Fosterville pasture establishment trial site.

Horizon	Description
A1 0-200 mm	dark reddish brown (5YR2.5/2) sandy clay loam common medium pebbles, few sub-rounded cobbles abundant fine roots
B1 200-280 mm	dark reddish brown (5YR3/2) medium clay common small pebbles, few sub-rounded cobbles many fine roots
B2 >280 mm	strong brown (7.5YR5/6) medium clay red to rust coloured mottles, quite gritty, very fine pebbles few fine roots

Note: Soil description based on definitions and terminology of land surface (McDonald *et al.* 2009) and soil profile (McDonald and Isbell 2009), and soil colour characterisation keys of Munsell soil color charts (Munsell 1973). Soil excavation ceased at 350 mm.

The experimental site was established by firstly spraying an application of Roundup® 460CT (a.i glyphosate 460 g/L) at a rate of 3 L/ha with Spraymate® Activator® surfactant (a.i. 900 g/L non-ionic surfactant) at a rate of 100 mL/100 L in August 2008 to eradicate the existing pasture. All plots were direct drilled with an Orjyord precision cone seeder on 17th August 2008 at the seeding rates given in Table 6.3. Irrigation was applied in three applications of 5 mm within a two week period to encourage germination and seedling vigour. The experimental site was surrounded with a 6 strand plain wire sheep exclusion fence allowing access of wildlife to area while excluding grazing by sheep. Agritone®750 (a.i. MCPA present as dimethylamine salt 750 g/L) at a rate of 1 L/ha with Spraymate® Activator® surfactant (a.i. 900 g/L non-ionic surfactant) at a rate of 100 mL/100 L was used to control broadleaf weeds in December 2008 and June 2009. Larger thistles were also removed by hand in January 2009. Fertiliser (NPKS 0-6-17-7) was applied in June 2009 at a rate of 140 kg/ha.

Mt. Pleasant site

A second experiment was undertaken within a 0.85 ha enclosure containing an established breeding population of wallabies and pademelons at DPIPWE's Mt. Pleasant Laboratories site in Launceston, Tasmania (41.3°S, 147.1°E). Four wallabies and 15 pademelons were recorded in a count of the Mt. Pleasant wallaby compound in spring 2008. Further counts in 2009 confirmed that these numbers remained steady throughout the study term. Wildlife freely accessed and were observed grazing experimental plots, wallabies occasionally during the day, but mostly at night. Pademelons grazed the experimental plots almost solely at night.

Black peppermint, white gum and prickly box dotted the grassland dominated by ryegrass, browntop (*Agrostis capillaris*), and clumps of phalaris, rushes and sedge. A detailed description of the upper soil profile is provided in Table 6.2. The soil fertility of the experimental site prior to commencing was as follows: Olsen P 20.7 mg/kg, Colwell K 327 mg/kg, mono-calcium phosphate extracted S 11.8 mg/kg, pH (H_20) 6.4 and electrical conductivity 0.110 dS/m.

The experimental site was established by firstly spraying an application of Roundup® 460CT (a.i. glyphosate 460 g/L) at a rate of 3 L/ha in May 2008 to eradicate the existing pasture. The area was left in a fallow for four months before being shallowly cultivated to remove the thick mat of organic matter remaining. The organic matter was raked off before a further application of Roundup® 460CT in October prior to direct drilling on the 24th October 2008. Agritone®750 (a.i. MCPA present as dimethylamine salt 750 g/L) was applied in early December 2008 at a rate of 0.5 L/ha, and again in late Dec 2008 (1 L/ha). Further applications of Agritone®750 were applied in March 2009 (0.5 L/ha) and in June 2009 (1 L/ha) to control broadleaf weeds. Irrigation was applied in three applications of 5mm over a two week period in October/November to encourage germination and seedling vigour. Fertiliser (NPKS 0-6-17-7) was applied in June 2009 at a rate of 100 kg/ha.

Table 6.2 Soil description of Mt. Pleasant pasture establishment trial site

Horizon	Description
A1 0-100 mm	dark brown (10YR3/3) sandy clay loam few fine pebbles abundant fine roots
B1 100-200 mm	yellowish brown (10YR5/4) light clay common medium pebbles, red to strong brown coloured mottles common fine roots
B2 >200 mm	dark brown (10YR3/3) light clay quite gritty with common small pebbles rare fine root

Note: Soil description based on definitions and terminology of land surface (McDonald *et al.* 2009) and soil profile (McDonald and Isbell 2009), and soil colour characterisation keys of Munsell soil colour charts (Munsell 1973). Soil excavation ceased at 450 mm.

6.2.2 Experimental design

Fosterville

A split plot experiment in a randomised complete block design (RCBD) consisting of three blocks was implemented (Appendix 2) to assess the effect of wildlife grazing on different pasture types during establishment. There were two main plot treatments; exposed to wildlife grazing, and exclusion of wildlife grazing during establishment, from here on referred to as unfenced and fenced, respectively. Each main plot was 6 m x 15 m and exclusion of grazing wildlife was achieved using 1.8 m high fencing (Figure 6.1). The fence was constructed using 2.4 m star pickets with Wallaby Wire® (110-8500-150 mm) around the base and Stocksafe® Longlife wire (110-900-150 mm) attached to the top to increase the height. The sub-plot treatments consisted of four different pasture types based on cocksfoot, ryegrass, phalaris and a mixture of grasses (Table 6.3). The sub-plots were 1.5 m x 15 m in size. Within each sub-plot, four plots were marked for pasture measurements. The experiment was positioned in the centre of the 1.0 ha experimental area with the remaining area sown with the ‘Mix’ seed combination (see Table 6.3).

It was unknown if the wildlife grazing surrounding exclusion cages influences the pasture growth within the exclusion cages or whether the exclusion cages themselves have a microclimate effect on the pasture within it. Fletcher (2006) suggested that quantifying the effect of the cage itself is necessary to adjust estimates of pasture lost to grazing. Having fenced and unfenced areas allowed us to evaluate the effect exclusion cages have on growth of the pasture within them. To assess the influence of exclusion cages on the growth of pasture within, four smaller (0.5 m x 0.5 m x 0.5 m) exclusion cages were randomly placed within each pasture type subplot, lying within each main treatment plot (Figure 6.1).

Mt. Pleasant

The Mt. Pleasant site followed the same treatments and split plot RCBD experimental design as the Fosterville site previously mentioned. Sowing rates were increased for the Mt. Pleasant sowing in an attempt to ensure sufficient numbers of plants germinated in the increasingly poor establishment conditions created by lack of rain (Table 6.4). Due to space limitations, only a 1.5 m buffer was sown around the experimental plots and the remaining area of the enclosure remained grassed.

Figure 6.1 Fosterville experimental site post sowing and implication of fencing and exclusion cage treatments. Fenced treatments are in the foreground and diagonally behind it. Unfenced treatments are less obvious to the left and right of the fenced treatment in the foreground.

Table 6.3 Description of seeding rate and treatments sown at Fosterville.

	Treatment seeding rate (kg/ha)			
Species	Mix	Phal./Sub	Cocks./Sub	Rye./Sub
Phalaris *Phalaris aquatica L.* cv. Australian	9	10	0	0
Cocksfoot *Dactylis glomerata L.*				
cv. Porto	2	0	3	0
Ryegrass *Lolium perenne L.*				
cv. Victoca	0.5	0	0	10
Sub Clover *Trifolium subterraneum L.*				
cv. Gosse	4	4	4	4
cv. Goulburn	4	4	4	4
cv. Karridale	4	4	4	4

Table 6.4 Description of seeding rate and treatments sown at Mt. Pleasant.

	Treatment seeding rate (kg/ha)			
Species	Mix	Phal./Sub	Cocks./Sub	Rye./Sub
Phalaris *Phalaris aquatica L.* cv. Australian	30	40	0	0
Cocksfoot *Dactylis glomerata L.* cv. Porto	6	0	40	0
Ryegrass *Lolium perenne L.* cv. Victoca	4	0	0	40
Sub Clover *Trifolium subterraneum L.*				
cv. Gosse	4	4	4	4
cv. Goulburn	4	4	4	4
cv. Karridale	4	4	4	4

6.2.3 Pasture measurements

Biomass accumulation

Pasture growth and biomass accumulation were measured by periodically harvesting the plots to a height of 40 mm to mimic grazing when the majority of plots in the fenced areas had reached 100-200 mm in height (Figure 6.2). Plots were cut with manual or mechanical hand shears and a consistent cutting height was achieved by using the top of the quadrat as a height guide. Samples were placed in plastic bags, tied and stored in a food cooler in the field to reduce respiration and water loss. Samples were oven dried at 60 °C for 48 hours and weighed to give a DM yield. Following harvest, the plots were mowed to a height of 40 mm with a rotary mower.

Figure 6.2 Harvesting one of the unfenced caged plots alongside an uncaged plot with battery powered mechanical hand shears. Note the pasture growth in the fenced treatment in the background.

Botanical composition

Botanical composition was estimated visually prior to harvest. The estimate was divided up into each species present and how much the canopy of each species covered the bare ground. These were recorded as a percentage along with the percentage of bare ground/residue pasture remaining. This method was adapted from a method for canopy cover discussed by t' Mannetje (2000b).

Wildlife numbers

Resources did not allow a measure of animal densities at Fosterville or Mt. Pleasant. Some comment of the species present and indicative number observed is outlined in

Section 6.2.1. If resources had allowed, a faecal pellet survey at regular time intervals within the Fosterville and Mt. Pleasant sites would have been employed.

6.2.4 Data analysis

Effect of fencing on pasture growth rates and yield

To assess the effects of wildlife grazing on the yield of the four pasture types, data from fenced and unfenced plots nested within each pasture sub-plot were pooled and fitted to a linear mixed model using SAS (version 9.1) with block added as a random term, and fence and pasture type and fence by pasture type terms treated as fixed effects. Fence by block was the random term used to test the fence effect, while the other fixed effects were tested with the residual error. This test was performed for each sample date. Fisher's LSD Post Hoc test was used to compare treatment means.

Effect of fencing on botanical composition

Botanical composition was summarised using the mean of the four exposed (uncaged) plots in each sub-plot treatment in fenced and unfenced main plots.

Effect of exclusion cages on growth rates and yield

To assess the influence of exclusion cages on pasture growth, exclusion cages were placed in both fenced and unfenced plots. The data from caged plots within fenced and unfenced plots within each pasture sub-plot were pooled and fitted to a linear mixed model using SAS (version 9.1) with fence and pasture type and fence by pasture type terms treated as fixed effects. Fence by block was the random term used to test the fence effect, while the other fixed effects were tested with the residual error. This test was performed for each sample date. Fisher's LSD Post Hoc test was used to compare treatment means.

6.3 Results

6.3.1 Influence of wildlife grazing during establishment on pasture production

6.3.1.1 *Fosterville*

At Fosterville, ANOVA tests of pasture growth rates found a significant ($P<0.05$) fencing treatment by pasture type interaction at all sampling dates except December 2009 (Table 6.5, Figure 6.3). There was a significant fencing treatment effect on dates in March ($F_{1, 2} = 147.69$; $P = 0.007$, July ($F_{1,2} = 90.86$; $P = 0.011$) and September ($F_{1,2} = 63.66$; $P = 0.0153$). There was a significant ($P<0.05$) pasture type effect on all dates except December ($F_{3, 12} = 3.20$; $P = 0.062$). Growth rates for mix and phalaris pasture types were greater in fenced treatments than unfenced treatments at harvest dates in March, July, and both September harvests, but not in October and December when the growth rates for mix and phalaris pasture types unfenced were greater than fenced. Growth rates of unfenced pasture types ryegrass and cocksfoot remained low (<20 kg DM/ha.day) until October 2009.

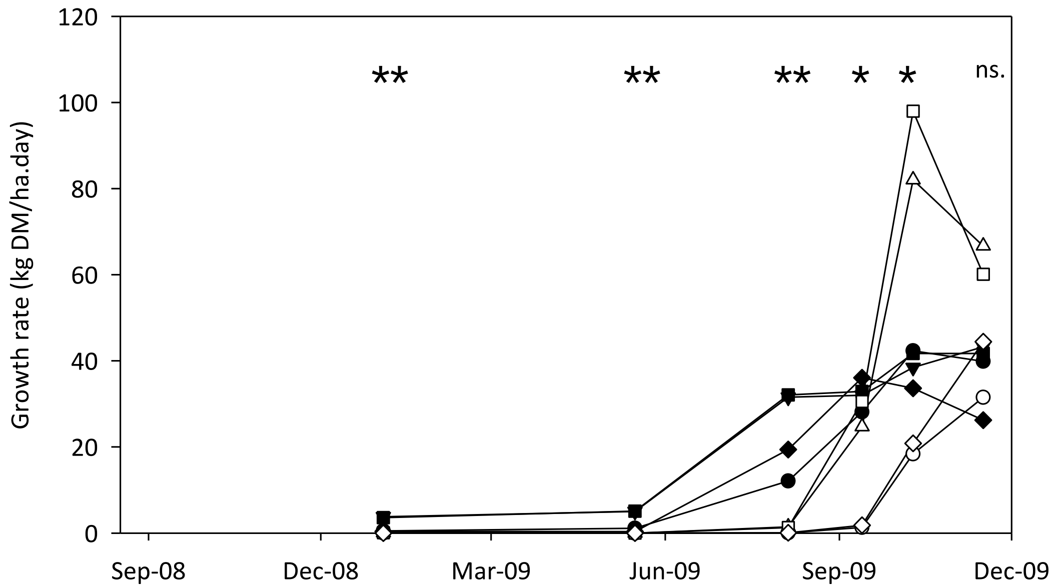


Figure 6.3 Growth rates (kg DM/ha.day) of the plots nested within each pasture sub-plot treatment at Fosterville. Fenced pasture types are cocksfoot (●), mix (▼), phalaris (■), and ryegrass (♦). Unfenced treatments have the same markers, but no fill. For each time interval significant pasture type by fencing treatment interaction is indicated by *(P<0.05) and ** (<0.01). Markers show growth rates at the midpoint of the growing period.

Table 6.5 Effects of fencing and pasture type on the growth rates of pasture at 6 harvest times at Fosterville.

	Value	Mar	July	Sept	Sept	Oct	Dec
Fencing (main plot treatment	$F_{1,2}$	147.69	90.86	63.66	12.27	17.27	3.92
	P	0.007	0.011	0.015	0.073	0.0533	0.186
Pasture type (sub-plot treatment)	$F_{3,12}$	8.25	7.10	15.45	4.35	5.75	3.20
	P	0.003	0.005	<0.001	0.027	0.0112	0.062
Fence*Pasture type interaction	$F_{3,12}$	7.66	7.10	11.53	4.04	5.07	1.60
	P	0.004	0.005	0.001	0.034	0.0170	0.242

When analysed as the total accumulated biomass, there was no significant (P>0.05) fencing treatment by pasture type interaction detected (Figure 6.4). There was, however, a significant (P = 0.042) fencing effect and a significant (P<0.001) pasture type effect on total accumulated biomass. In fenced and unfenced treatments, both mix and phalaris pasture types had a significantly (P<0.05) higher biomass compared to cocksfoot and ryegrass pasture types.

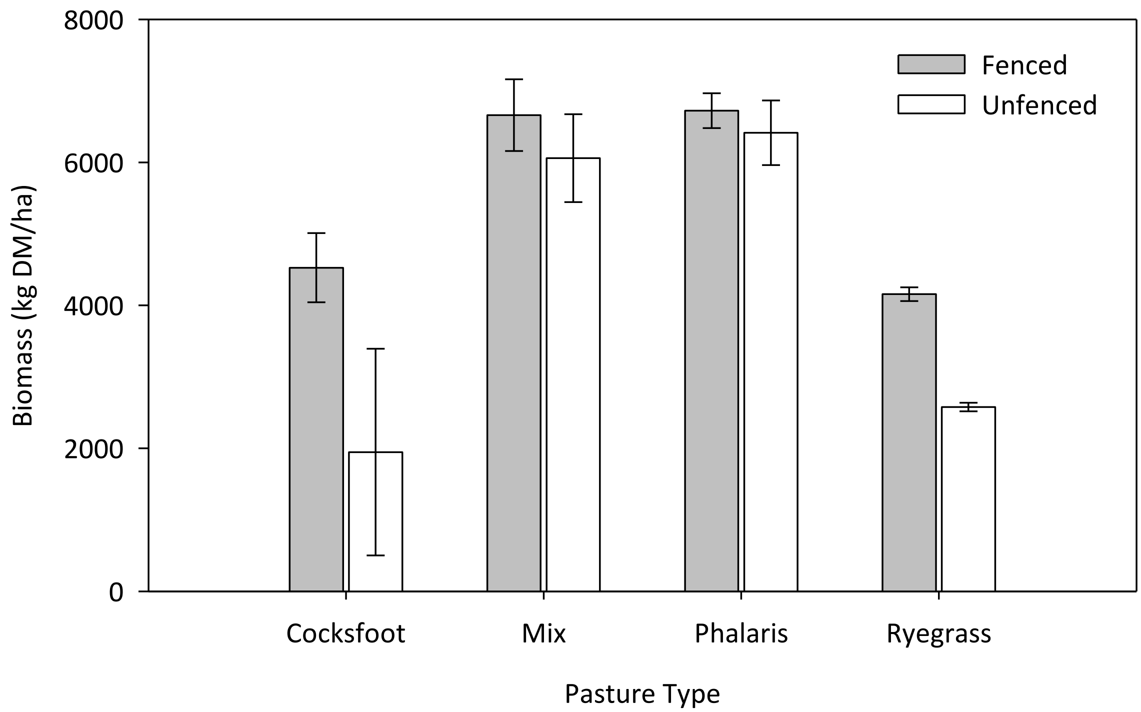


Figure 6.4 Mean total accumulated biomass of the plots nest within each pasture sub-plot treatment of pasture types (cocksfoot, mix, phalaris, and ryegrass) in fenced and un-fenced treatments at Fosterville. Error bars represent the standard error of the mean.

6.3.1.2 *Mt. Pleasant*

At Mt. Pleasant, an ANOVA test of growth rates indicated a significant (P<0.05) fencing treatment by pasture type interaction was recorded on all dates except January ($F_{3,12}$ = 0.94; P = 0.453) and March ($F_{3,12}$ = 3.23; P = 0.061) (Table 6.6, Figure 6.5). Higher growth rates were recorded in fenced treatments than unfenced treatments. There was no significant (P>0.05) fencing treatment effect at any dates except March ($F_{1,2}$= 19.57; P = 0.048). There was no significant (P>0.05) pasture type effect at any dates except October ($F_{3,12}$ = 5.96; P = 0.010) and November ($F_{3,12}$ = 8.78;P = 0.002).

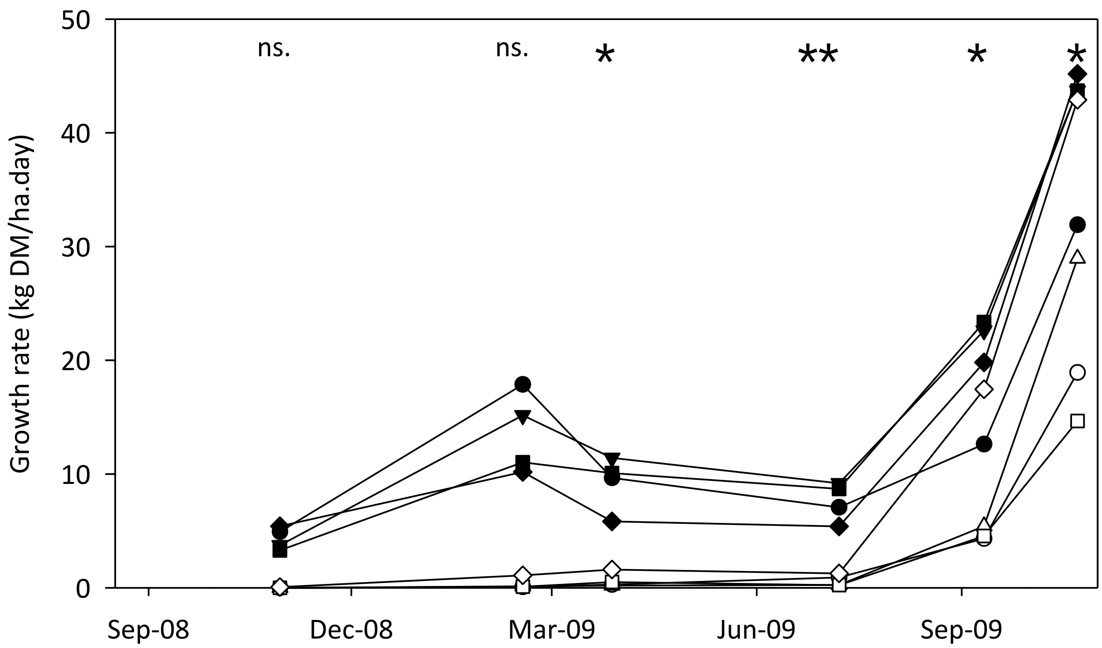


Figure 6.5 Growth rates (kg DM/ha.day) of the plots nested within each pasture sub-plot treatment at Mt. Pleasant. Fenced pasture types are cocksfoot (●), mix (▼), phalaris (■), and ryegrass (♦). Unfenced treatments have the same markers, but no fill. For each time interval significant pasture type by fencing treatment interaction is indicated by *(P<0.05) and ** (<0.01). Markers show growth rates at the midpoint of the growing period.

Table 6.6 Effects of fencing and pasture type on the growth rates of pasture at 6 harvest times at Mt. Pleasant.

	Value	Jan	Mar	May	Aug	Oct	Nov
Fencing (main plot treatment)	$F_{1,2}$	10.97	19.57	8.17	5.34	15.07	3.38
	P	0.080	0.048	0.104	0.147	0.060	0.207
Pasture types (sub-plot treatment)	$F_{3,12}$	1.03	2.30	1.54	1.92	5.96	8.78
	P	0.412	0.129	0.254	0.180	0.010	0.002
Fencing *Pasture type interaction	$F_{3,12}$	0.94	3.23	4.46	6.40	5.16	3.92
	P	0.453	0.061	0.025	0.008	0.016	0.037

When analysed as the total accumulated biomass, there was a significant fencing treatment effect (P = 0.051) and a significant pasture type effect (P = 0.032) on total accumulated biomass (Figure 6.6). There was also a significant fencing treatment by pasture type interaction (P = 0.004). This interaction was caused by the relatively high yield in the ryegrass treatment compared with the yield of the other three pasture types, when unfenced.

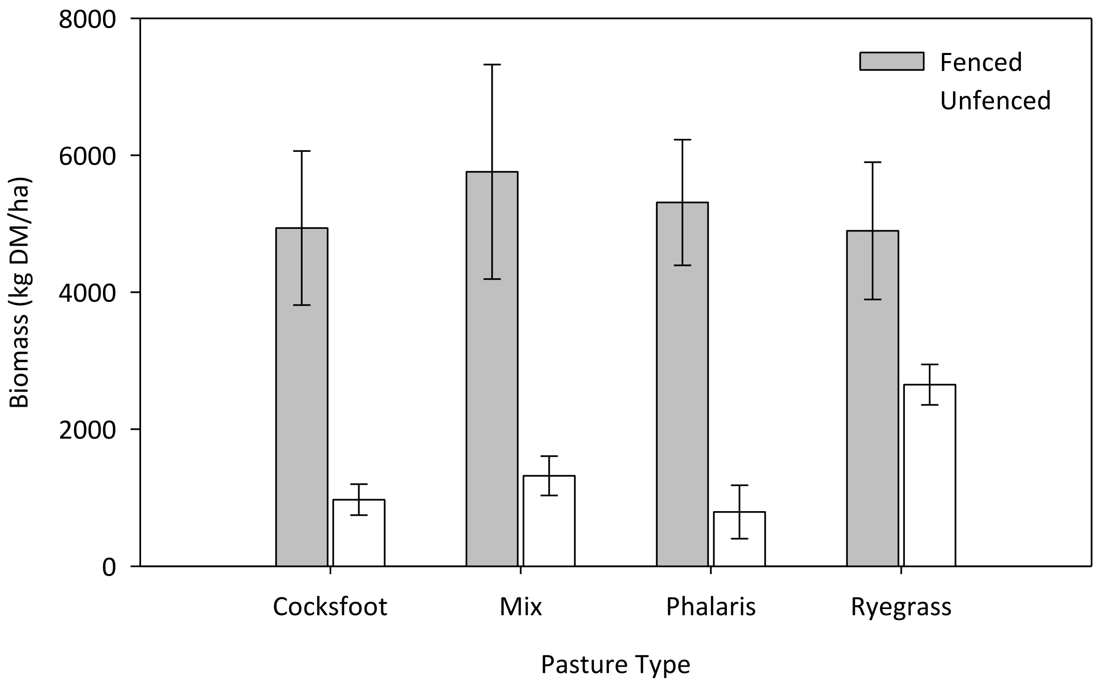


Figure 6.6 Mean total accumulated biomass of the plots nested within each pasture sub-plot treatment of pasture types (cocksfoot, mix, phalaris, and ryegrass) in fenced and unfenced treatments at Mt. Pleasant. Error bars represent the standard error of the mean.

6.3.2 Influence of wildlife grazing during establishment on botanical composition

6.3.2.1 *Fosterville*

This results section describes the changes in botanical composition observed and recorded between April 2009 and December 2009 following sowing in August 2008. Only data from exposed (uncaged) plots were used for these observations. Data from the four exposed plots nested within the pasture type sub-plots were pooled and averaged.

Plant cover was similar in fenced and unfenced treatments at the initial observation (Figure 6.7). The amount of bare ground was initially high (>60%) in all treatments. Bare ground was initially higher in fenced (73%) than unfenced (60%) phalaris treatments. However, bare ground was similar between fenced and unfenced plots in ryegrass, cocksfoot and mix pasture type treatments. Bare ground decreased significantly in all treatments over the study period. The amount of bare ground was lowest in fenced treatments earlier in the growing season (September) compared with unfenced treatments before increasing again through until December.

The major sown grass in each pasture type increased in composition in both main plot treatments. For example, phalaris increased from 27% to 69% with a peak of 87% in unfenced treatments and 23% to 46% with a peak of 87% in fenced treatments. Phalaris in fenced mixed treatments reached 88%, but decreased to 50% by December.

Composition of cocksfoot peaked in September at 35% and 67% in the unfenced and fenced treatments, respectively. The composition of the major sown grass peaked higher in fenced than unfenced treatments, with the exception of the phalaris pasture type where they were identical. In December the major sown grass was greater in unfenced treatments than fenced with the exception of cocksfoot in cocksfoot pasture types.

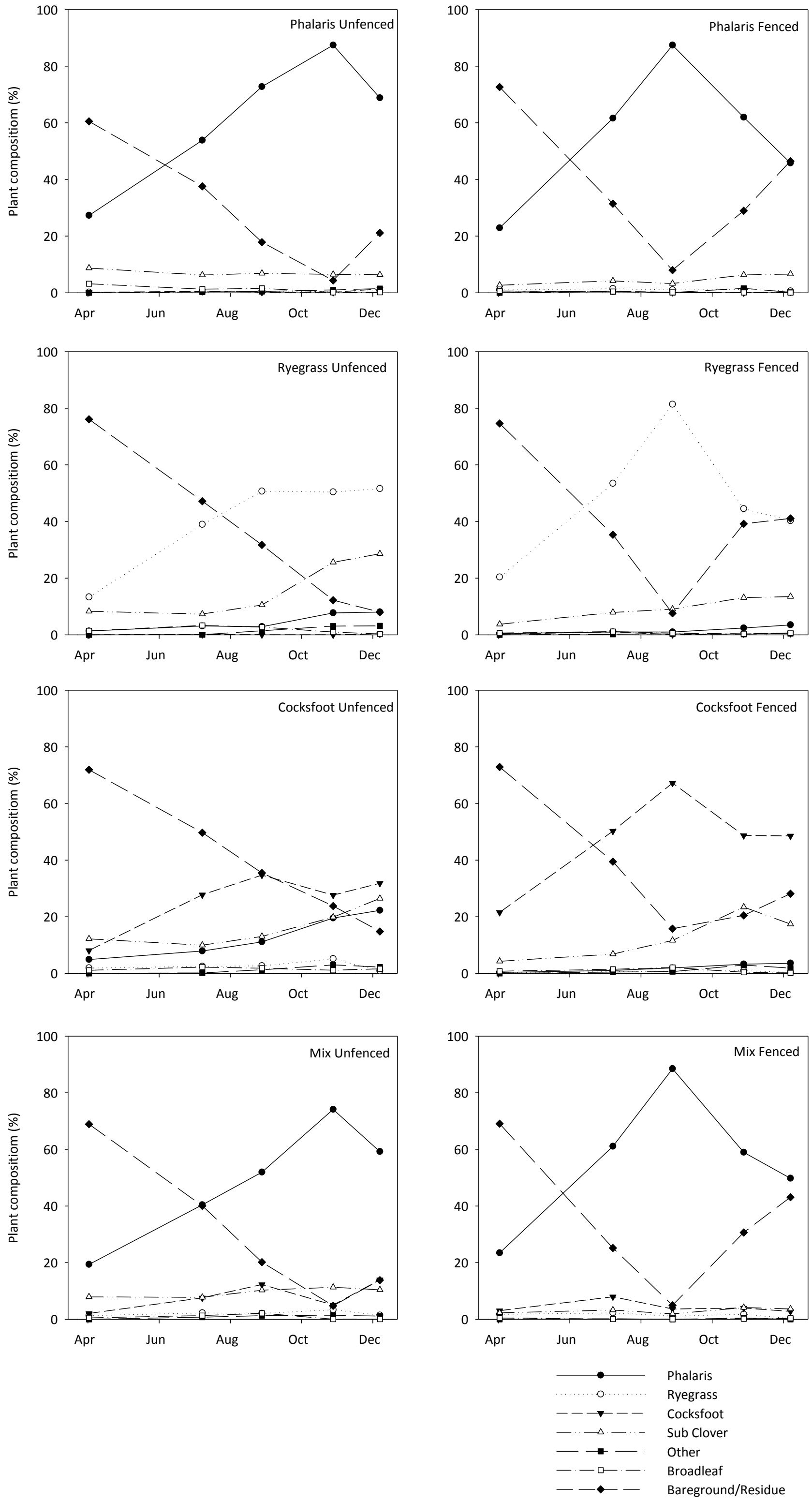


Figure 6.7 Mean botanical composition of four exposed (uncaged) plots nested within each pasture type sub-plot at five observations between April 2009 and December 2009 at Fosterville. Each marker represents a survey date.

The composition of subterranean clover increased over the trial period. Higher levels were recorded in unfenced treatments, except phalaris pasture type where levels were similar. Subterranean clover in the unfenced ryegrass treatment was 29% compared with 14% in fenced and in the cocksfoot treatment was 26% compared with 17%. There was no increase in the cover of unsown grasses.

6.3.2.2 *Mt. Pleasant*

This results section describes the changes in botanical composition observed and recorded between January 2009 and November 2009 following sowing in October 2008. Only data from exposed (uncaged) plots were used for these observations. Data from the four exposed plots nested within the pasture type sub-plots were pooled and averaged.

The most notable difference from the first observation was that plant cover was significantly greater in fenced treatments than in unfenced treatments (Figure 6.8). The amount of bare ground in unfenced treatments was initially high (>60%) in January. This decreased in all pasture types over the trial period to below 30% with the bare ground in the mix pasture type unfenced treatment decreasing from 84% to 18%. In contrast, bare ground was 20% or less in fenced treatments in January and if anything slightly increased over the trial period, but remained less than 30% in all pasture types in November 2009.

The composition of ryegrass increased in all pasture types in unfenced treatments. Initially, the composition of ryegrass in the pasture was 5% or less in unfenced phalaris, cocksfoot and mix pasture types, but increased by November to greater than 40% in all cases. In the unfenced ryegrass pasture type, the composition of ryegrass increased from 31% to 74% in November, peaking at 82% in May. In contrast, levels of ryegrass remained low (<10%) in November in phalaris, cocksfoot and mix fenced treatments. In addition, ryegrass decreased from 71% to 45% in the fenced ryegrass pasture type.

The major sown grass species in each pasture type increased in composition over the study period in unfenced treatments. For example, phalaris increased from 6% to 25% in

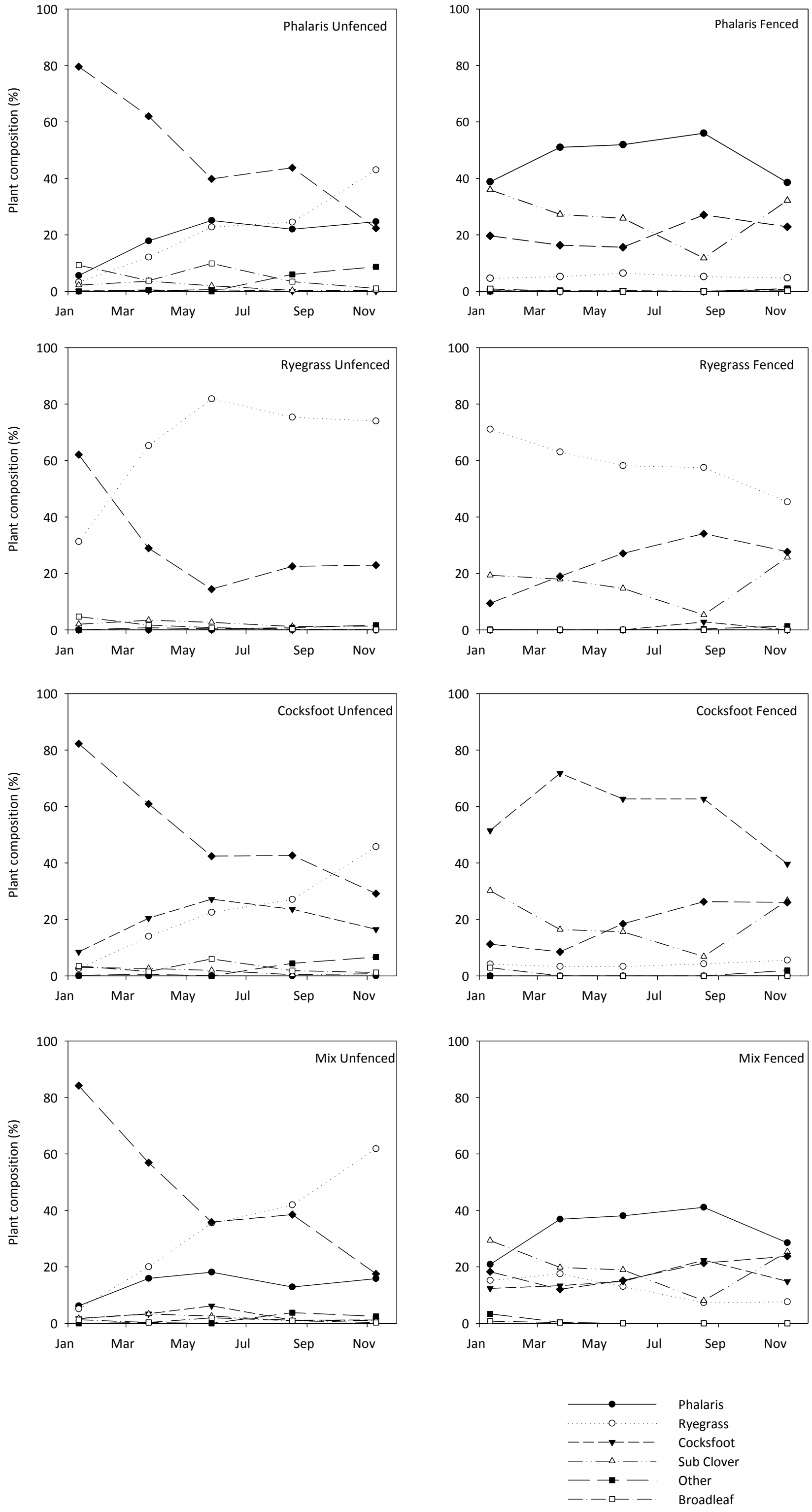


Figure 6.8 Mean botanical composition of four exposed (uncaged) plots nested within each pasture type sub-plot at five observations between January 2009 and November 2009 at Mt. Pleasant. Each marker represents a survey date.

unfenced phalaris treatments and cocksfoot increased from 9% to 17%, but peaked at 27% in unfenced cocksfoot treatments. The composition of the major sown species in each pasture type in fenced treatments was initially considerably higher than in unfenced treatments. The major sown species increased in fenced mix treatments (21% to 29%), remained stable in fenced phalaris treatments (39% to 39%), but decreased in fenced cocksfoot (52% to 40%) and fenced ryegrass (71% to 45%) treatments.

6.3.3 Evaluation of influences of exclusion cages on pasture production

6.3.3.1 *Fosterville*

The presence of exclusion cages *per se* may have an effect on pasture growth and production since the cages may change the microenvironment normally experience by pasture species. To evaluate the potential influence of exclusion cages on pasture growth rates, the data from the caged plots within fenced and non-fenced treatments were analysed. It was hypothesised that the growth rate of pasture in the exclosure cages in the non-fenced treatments could be higher since grazing by wildlife would reduce the above ground biomass of the pasture immediately outside of the cages and this may increase the availability of light, moisture and soil nutrients to the pasture protected from grazing within the cages.

The data from the *in situ* experiments at Fosterville appear consistent with the hypothesis that the exclusion cages may influence the growth rate of the pastures they are protecting from wildlife grazing under some circumstances. There was a significant ($P<0.05$) fencing treatment effect on growth rates in September, October and December with higher growth rates in unfenced treatments than fenced (Table 6.7). When averaged across pasture types, growth rates were 77.5 kg DM/ha per day in unfenced plots and 33.9 kg DM/ha per day in fenced plots in October and 66.3 kg DM/ha per day in unfenced and 38.0 kg DM/ha per day in fenced plots in December (Figure 6.9).

Similarly, when analysed as the total accumulated biomass (growing period 17-8-2008 to 7-12-2009), there was a significant ($P = 0.011$) fencing treatment effect, with pasture in the unfenced treatments yielding a higher biomass than those in fenced treatments (Figure 6.10). When averaged across pasture types, unfenced plots yielded 9,754 kg DM/ha compared with 6,442 kg DM/ha in fenced plots.

No significant ($P>0.05$) fencing treatment by pasture type interaction was found on pasture growth on any dates. There was a significant ($P<0.05$) pasture type effect on growth rates at all dates except July and December. For example, when averaged across fencing treatment plots, phalaris (65.6 kg DM/ha per day) had higher pasture growth rates than cocksfoot (42.9 kg DM/ha per day) for the October harvest.

When analysed as the total accumulated biomass, no significant ($P>0.05$) fencing treatment by pasture type interaction was detected at Fosterville (Figure 6.10). There was a significant ($P<0.05$) pasture type effect, with phalaris (8,582 kg DM/ha) and Mix (9172 kg DM/ha) pasture types yielding greater than cocksfoot (7,548 kg DM/ha) and ryegrass (7,091 kg DM/ha).

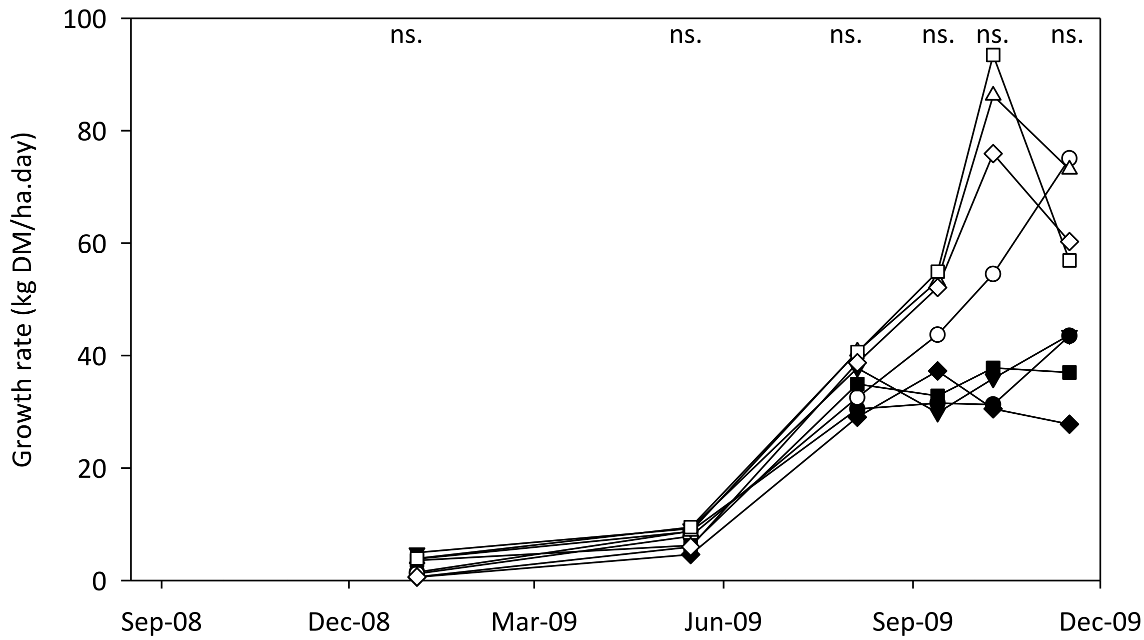


Figure 6.9 Growth rates (kg DM/ha.day) of the caged plots nested within each pasture sub-plot treatment at Fosterville. Fenced pasture types are cocksfoot (●), mix (▼), phalaris (■), and ryegrass (♦). Un-fenced treatments have the same markers, but no fill. No significant (P>0.05) pasture type by fencing treatment was found.

Table 6.7 Effects of fencing and pasture type on the growth rates of pasture at 6 harvest times at Fosterville. Data from caged plots were used to investigate the influence of cages on pasture growth.

	Value	**Mar**	**Jul**	**Sept**	**Sept**	**Oct**	**Dec**
Fencing (main plot treatment)	$F_{1,2}$	0.37	0.35	18.01	19.66	84.52	48.62
	P	0.605	0.613	0.051	0.047	0.012	0.020
Pasture type (sub-plot treatment)	$F_{3,12}$	27.97	3.17	3.69	4.00	6.51	3.10
	P	<0.001	0.064	0.043	0.035	0.007	0.067
Fencing * Pasture type interaction	$F_{3,12}$	0.81	1.13	0.86	3.02	3.32	0.42
	P	0.513	0.377	0.487	0.072	0.057	0.740

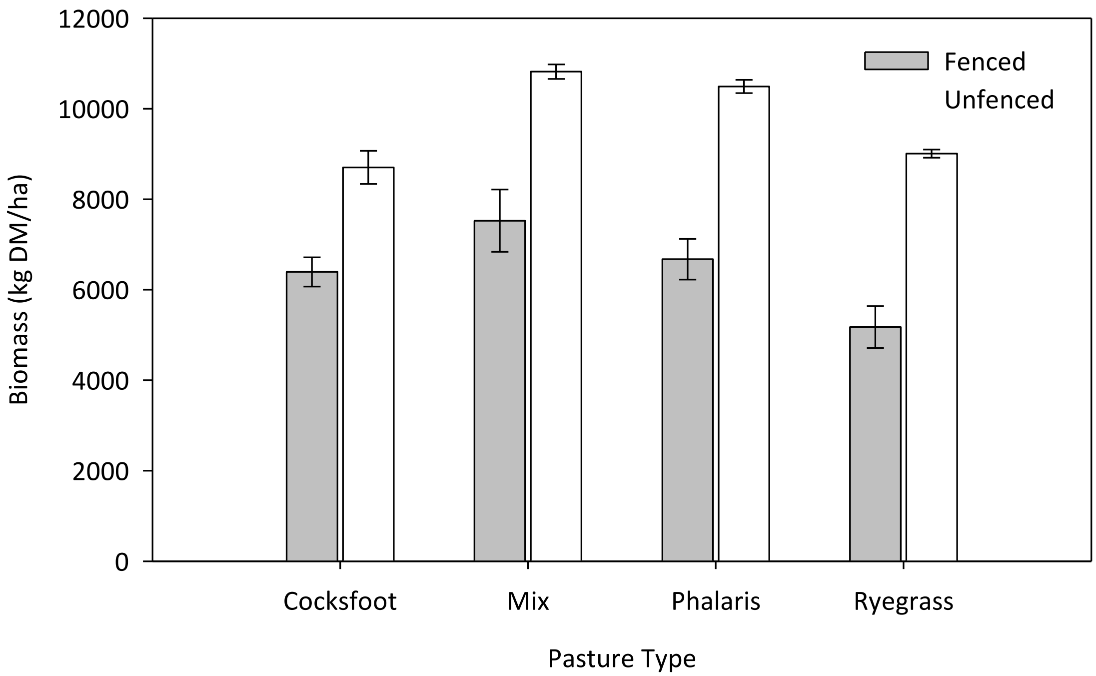


Figure 6.10 Mean total accumulated biomass of the caged plots nested within each pasture sub-plot treatment of pasture types (cocksfoot, mix, phalaris, and ryegrass) in fenced and un-fenced treatments at Fosterville. Error bars represent the standard error of the mean.

6.3.3.2 Mt. Pleasant

Data from the *ex situ* research at Mt. Pleasant was also analysed to evaluate the potential influence of exclusion cages on pasture growth. Similar to Fosterville, a significant fencing treatment effect on growth rates was found on some harvest dates, including in March ($F_{1,2} = 103.4$; $P = 0.010$) and May ($F_{1,2} = 34.05$; $P = 0.028$) with higher growth rates in unfenced treatments than fenced (Figure 6.11). When averaged across pasture types for the harvest dates of March and May, unfenced plots produced pasture growth rates of 20.9 kg DM/ha per day and 14.9 kg DM/ha per day compared with fenced plots producing 14.1 kg DM/ha per day and 9.1 kg DM/ha per day, respectively.

However, in contrast to Fosterville, no significant ($P>0.05$) relationship was detected between total accumulated pasture (growing period 24-10-2008 to 10-11-2009) biomass and fencing treatment at Mt. Pleasant (Figure 6.12). This was probably due to higher ryegrass yields occurring in fenced plots at Mt Pleasant (5,126 kg DM/ha) compared to unfenced plots (4,393 kg DM/ha).

Also in contrast to Fosterville, a significant fencing treatment by pasture type interaction was found for growth rates at Mt. Pleasant at two harvest dates, March ($F_{3,12} = 6.97$; P = 0.006) and May ($F_{3,12} = 6.51$; P = 0.007) (Table 6.8, Figure 6.11). There was a significant (P<0.05) pasture type effect on growth rates at all dates except January. When averaged across main plots the phalaris pasture type produced higher pasture growth rates at 25.0 kg DM/ha per day than Mix (21.6 kg DM/ha per day), cocksfoot (12.9 kg DM/ha per day) and ryegrass (16.6 kg DM/ha per day) in October 2009.

A significant (P<0.001) relationship was found between total accumulated biomass and pasture type. When averaged across main plots, phalaris (6,904 kg DM/ha) yielded higher than mix (6,438 kg DM/ha), cocksfoot (5,297 kg DM/ha) and ryegrass (4,759 kg DM/ha). There was also a significant (P = 0.002) fencing treatment by pasture type interaction (Table 6.8, Figure 6.11). This pattern may also be explained by the ryegrass pasture type producing higher yields in the fenced compared to unfenced main plots.

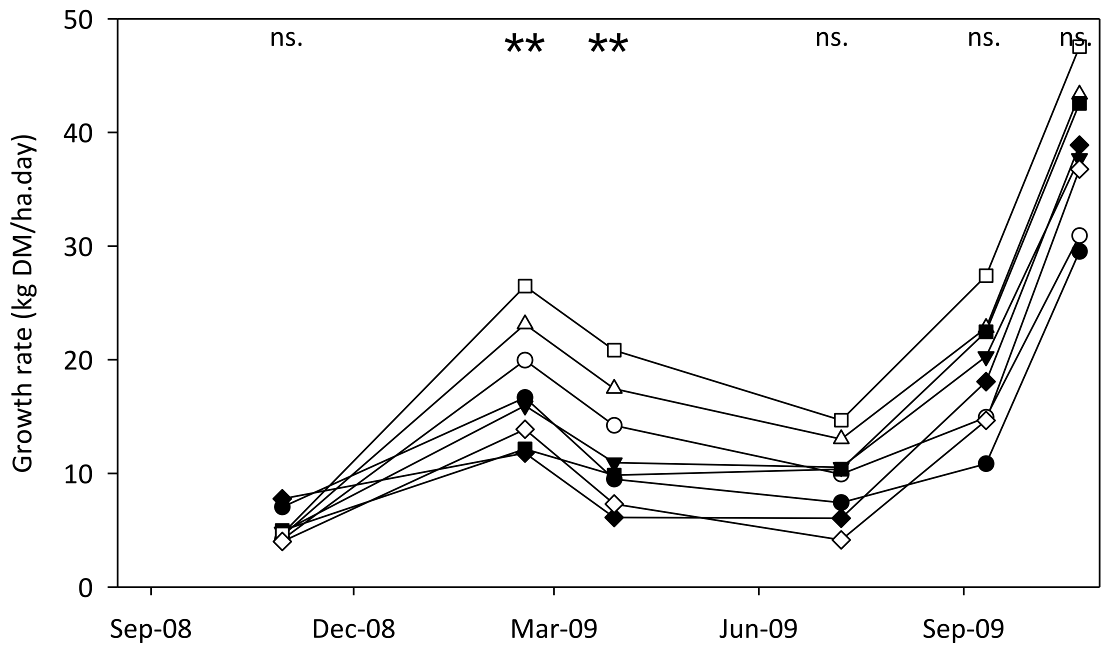


Figure 6.11 Growth rates (kg DM/ha.day) of the caged plots nested within each pasture sub-plot treatment at Mt. Pleasant. Fenced pasture types are cocksfoot (●), mix (▼), phalaris (■), and ryegrass (♦). Un-fenced treatments have the same markers, but no fill. For each time interval significant pasture type by fencing treatment interaction is indicated by *(P<0.05) and ** (P<0.01).

Table 6.8 Effects of fencing and pasture type on the growth rates of pasture at 6 harvest times at Mt. Pleasant. Data from caged plots were used to investigate the influence of cages on pasture growth.

	Value	Jan	Mar	May	Aug	Oct	Nov
Fencing (main plot treatment)	$F_{1,2}$	2.66	103.09	34.05	7.21	0.34	0.12
	P	0.244	0.010	0.028	0.115	0.619	0.762
Pasture type (sub-plot treatment)	$F_{3,12}$	1.09	9.12	22.97	18.51	13.92	9.46
	P	0.390	0.002	<0.001	<0.001	<0.001	0.002
Fencing * pasture type interaction	$F_{3,12}$	2.42	6.97	6.51	2.81	1.73	0.78
	P	0.116	0.006	0.007	0.085	0.215	0.526

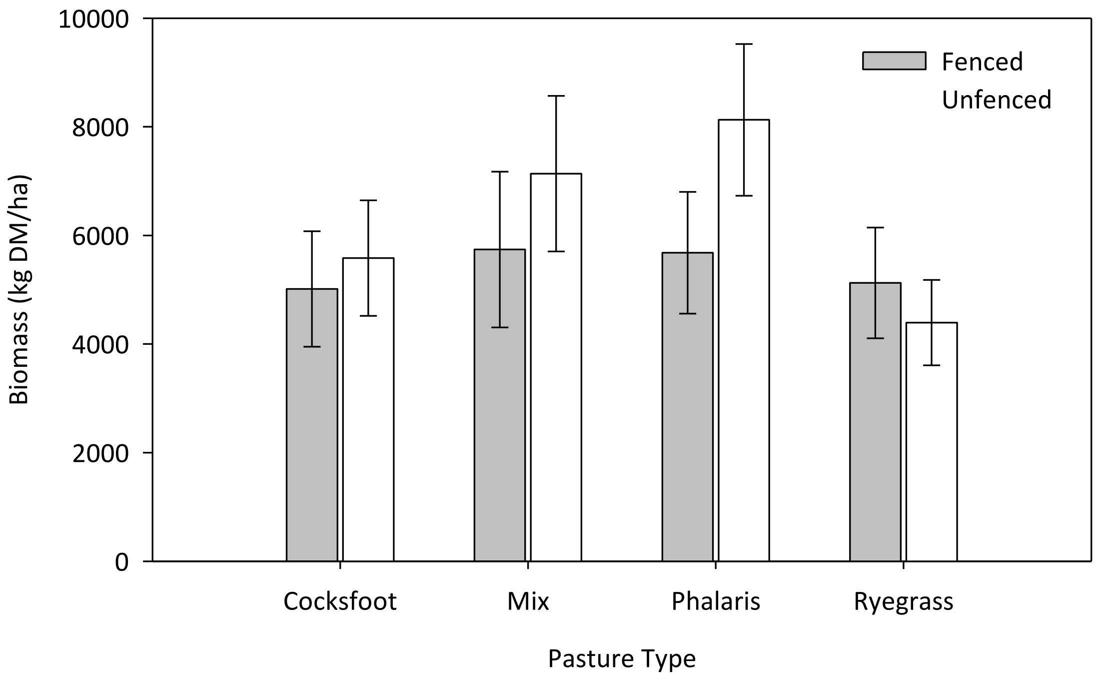


Figure 6.12 Mean total accumulated biomass of the caged plots nested within each pasture sub-plot treatment of pasture types (cocksfoot, mix, phalaris, and ryegrass) in fenced and unfenced treatments at Mt. Pleasant. Error bars represent the standard error of the mean.

6.4 Discussion

6.4.1 Influences of wildlife grazing during establishment on pasture production and botanical composition

Wildlife, in particular the macropod species have a preference for feeding on short pastures with high nitrogen content and digestibility (Johnson and Rose 1983; Dawson 1989; Jarman and Phillips 1989). Taylor (1985) compared kangaroo grazing on an improved and unimproved pasture in New South Wales and suggested that survival rates of juveniles was higher on the improved pasture, leading to a higher population density. Taylor (1985) also found that the animals ate less (per animal) of the higher quality feed on the improved pasture. Poorer feed quality in the form of lower digestibility leads to higher intakes on the unimproved pasture (Taylor 1985). Given a choice between the two pastures it would be expected that the wildlife would favour an improved pasture over an unimproved pasture for grazing. At Fosterville, the trial area was surrounded by unimproved native woodland vegetation and a poor quality pasture characterised by naturalised exotic annual grasses. These grasses tended to senesce earlier in the summer and as a result the trial area became a preferred feeding ground for wildlife as it retained short green feed for longer.

Grazing of establishing pastures by wildlife resulted in significant reductions in accumulated biomass harvested at Mt. Pleasant and to a lesser extent at Fosterville. At Mt. Pleasant there was no significant difference in biomass between all pasture types in fenced treatments, however in the unfenced treatments there was significantly more DM in the ryegrass pasture type than any of the other pasture types. This may indicate a preference for pasture species other than ryegrass. Given that much of the Mt. Pleasant wallaby compound is dominated by ryegrass wildlife may have been seeking variety in their diet. However, this cannot be proven by these results and would require an examination of faecal or gut material and survey of the entire pasture. A faecal pellet survey may have provided useful data in determining if there was a difference in wildlife population density and grazing pressure between Fosterville and Mt. Pleasant. Possible methods to perform this survey were outlined and implemented by Hill (1981a), Coulson and Raines (1985), Perry and Braysher (1986) and Johnson *et al*. (1987). Differing grazing densities may have provided reason for the difference in the reduction of biomass at the two sites. Limited resources prevented this type of survey from being undertaken.

Growth rates of unfenced phalaris and mix pasture types increased dramatically at Fosterville from September 2009 leading to greater production which resulted in little difference between fenced and unfenced treatments by the end of the trial. This was not the case at Mt. Pleasant. Feed availability increased in late 2009 at Fosterville as a result of heavy rains (See Table 3.4), which may have reduced the grazing pressure on trial plots. Unfenced treatments of phalaris and mix may have shown compensatory growth under reduced grazing pressure. In addition, fenced treatments may have entered a reproductive state earlier in the season due to reduced grazing pressure, lowering growth rates relative to unfenced plots.

There was no significant difference in biomass between all pasture types in fenced treatments at Mt. Pleasant. In contrast at Fosterville, the mix and phalaris pasture types produced significantly more DM than cocksfoot and ryegrass. Mix and phalaris pasture types are dominated by phalaris which is deep rooting and may be better suited to the lower rainfall climate at Fosterville (see Table 2.1). Phalaris is best suited to regions receiving a minimum of 400 mm average annual rainfall while ryegrass is better suited to a minimum of 500-700 mm (Knox *et al.* 2006). Differences in production of mix and phalaris pasture types between Mt. Pleasant and Fosterville may also be accounted for by the longer trial period at Fosterville (3 months longer).

Wildlife have been known to reduce the cover of grassland and pasture plants through grazing. For example, kangaroos were thought to reduce the cover of a number of species including *Themeda australis* and *Glycine clandestine* in a study by Neave and Tanton (1989) at Tidbinbilla Nature Reserve in the Australian Capital Territory. Furthermore, increasing rabbit numbers on Macquarie Island were implicated with a greater area of bare ground (Scott and Kirkpatrick 2008) and bare ground increased with higher rabbit density in a study by Croft *et al.* (2002). In a study by Bridle and Kirkpatrick (1999) subalpine vegetation on the Central Plateau of Tasmania, sheep were thought to have a greater impact on the cover of vegetation that the wildlife species present. In the current study, grazing by wildlife species has affected cover of individual plant species and the level of bare ground overall. Cover of the main sown grass in each pasture type was generally higher in fenced than unfenced treatments at both Mt. Pleasant and Fosterville. This indicates that controlling wildlife provides protection for the main grass species to better establish than if wildlife were uncontrolled. At Mt. Pleasant the level of bare ground was significantly higher at the beginning of the trial in unfenced treatments, but this level declined throughout the trial period. The bare ground was replaced mostly by ryegrass, growing as a voluntary weed species. This may have struck from seed reserves in the soil or from dormant plants unaffected by the applications of herbicide prior to sowing. Perennial ryegrass is known to be very vigorous and can tolerate heavy continuous grazing (Knox *et al.* 2006). In addition perennial ryegrass establishes more easily from seed than cocksfoot and phalaris (Knox *et al.* 2006). Ryegrass appeared to be less affected by wildlife grazing than the other sown grass species and as previously discussed may be a result of wildlife seeking variety in their diet. It may also reflect a superior ability of ryegrass in these conditions to withstand and recover from heavy grazing by wildlife than the other grass species. These results indicate that some plant species are favoured by grazing. A comparable study was indicated by Neave and Tanton (1989), who reported that *Bromus molliformis* increased in cover and *Danthonia* sp. increased in occurrence in the presence of kangaroo grazing.

At Fosterville the amount of bare ground was similar in fenced and unfenced treatments at the beginning of the surveys. This may indicate that the grazing pressure by wildlife was not as severe at Fosterville as at Mt. Pleasant, at least at the start of the trial period. Wildlife may have taken longer to find the establishing pasture at Fosterville or the experimental plots may have been buffered by the larger (1 ha) area sown. The amount of bare ground decreased over the trial period in both fenced and unfenced treatments as a result of re-colonisation by the main sown grass species in each of the pasture types.

The cover of subterranean clover did not appear to be affected by wildlife grazing at Fosterville but clearly was at Mt. Pleasant. Two possible contributing factors were identified. The cover of legumes by visual observation within the surrounding pastures at Fosterville was much greater than at Mt. Pleasant. Small amounts of previously absent legumes at Mt. Pleasant could be quickly grazed to a low level of cover and may be favoured to add variety and nutrition to the diet. In addition, the diet selection of pademelon (present at Mt. Pleasant but not Fosterville) has been shown to include more browse plants including legumes than the larger wallaby and kangaroo which prefer mainly grasses. Subterranean clover, being an annual legume, survives from one season to the next as seed. Therefore, the ability of plants to enter the reproductive phase and seed set is critical for the persistence of the species. The effect of heavy grazing by wildlife during the flowering and seed set phases of subterranean clover appears to be related to the pasture cultivar and the time of year (season). Defoliation during flowering has had both positive (Steiner and Grabe 1986; Conlan *et al.* 1994) and negative effects

(Rossiter 1961; Collins 1978; Archer 1990; Conlan *et al.* 1994) on seed yields. At Mt. Pleasant wildlife grazing has almost totally reduced the cover of subterranean clover leaving little chance for flowering and seed set. Therefore, it is possible that subterranean clover could quickly be removed from areas of establishing pasture, particularly where there are high numbers of pademelons. This finding would account for anecdotal reports of wildlife grazing the legume component out of pastures. These observations are consistent with Gregory (1989) who reported that a greater legume content was present within exclusion cages than within areas exposed to grazing by pademelons and wallabies. These results are similar to the effects of rabbits grazing on the subterranean clover component on pastures (Croft *et al.* 2002), which in turn may have led to reduced sheep production (Fleming *et al.* 2002). The level of bare ground at Fosterville appeared to vary more throughout the trial period and may be related to the more seasonal growing conditions.

6.4.2 Landscape context

The management of establishing pastures in the Midlands region is different from other, higher rainfall areas of Tasmania Following sowing in spring, the pasture is encouraged to mature and set seed by not grazing the pasture during the first spring and summer. This allows the new plants time to establish deep roots, build energy reserves and the extra seed provides a means of ongoing plant recruitment. A light grazing, at a high stocking rate of sheep in late summer/early autumn helps to shatter the seed heads and trample the seed into the soil. This process results in the removal of some of the rank material, but does not lead to overgrazing of the growing points. For this reason harvest cuts for biomass and DM estimates were not taken during spring and summer of the first season.

The establishment of large areas of pasture is likely to spread the impact of browsing damage over a greater area. This is dependent on the amount of pasture in close proximity to the vegetation edge. A pasture with a small interface may concentrate the browsing within a small area, while pasture with a larger interface may spread the damage over a larger area. This is particularly important when it comes to pasture establishment as heavy or light wildlife grazing may be the difference between life and death of a plant, and therefore part of the pasture.

One management strategy used in the first year of pasture establishment is to allow grasses to set seed, which is later important for plant recruitment. Red-necked wallabies consume a proportion of grass seed heads (Jarman and Phillips 1989) and as such may remove some of the recruitment potential of the establishing pasture, depending on digestion of the seed head in the gut. The continual grazing of pastures can also prevent pasture from going to seed. During the first summer after sowing it was noted that grasses in the exposed plots and surrounding buffer were prevented from setting seed by browsing. This may have limited plant production and plant recruitment in following seasons. It is also suggested by Campbell *et al* (1987) that spelling phalaris helps it become dormant and consequently improves the survivability of plants through their first summer. Given the dry conditions caused by below average rainfall in the period leading up to, and during summer 2008-09 it is reasonable to suggest that some plants were lost due to not entering dormancy. Following establishment, semi winter active cultivars of phalaris such as ‘Australian’ are very tolerant of continuous grazing (Lodge 1997).

6.4.3 Evaluation of grazing exclusion cages on pasture production

The use of exclusion cages in measuring pasture biomass has been outlined (Brown 1954; Carter 1962; t' Mannetje 1978; t' Mannetje 2000b) and is often referred to as the ‘difference method’ or the ‘cage technique’. This method has its limitations (Parsons *et al.* 1984), but remains one of the most commonly used methods for estimating production in grazed swards. Exclusion cages have also been the main method used in measuring DM loss to wildlife grazing in Tasmania (Statham and Rayner 1995; Donaghy and Tegg 2001) and most recently Smith *et al*. (2012). The size, shape and application have varied within these trials. Exclusion cages have also been used to measure pasture growth in dairy pastures (Nie *et al.* 2004) and to monitor the effects of grazing on pastures, for example sheep on *Astrebla* grasslands (Orr and Evenson 1991). Exclusion cages have been used in a number of other applications. For example, to evaluate the effect of wild fish on sediment build up from caged aquaculture (Felsing *et al.* 2005) and to assess the effect of predators on African bollworm *Helicoverpa armigera* numbers in cotton (van den Berg and Cock 1993).

Generally, exclusion cages are accepted as having little influence on the growth of the plants within them. However, methods involving exclusion cages have been criticised (Daubenmire 1940) due to the influence of cages on the internal environment and therefore on the growth of pastures. Williams (1951) stated that it was important to investigate the effects of exclusion cages on pasture growth due to their influence on the micro-environment. Exclusion cages were shown to reduce wind velocity by 26%, reduce light by 11%, and humidity was generally higher inside the cages than in uncovered pasture (Williams 1951). Studies have shown that exclusion cages can have both positive and negative effects on plant growth. Dry matter was increased by 11% in an un-grazed ryegrass/white clover pasture under exclusion cages (Cowlishaw 1955) and cages also had a positive effect on forage production in un-grazed true prairie vegetation in Kansas, USA (Owensby 1969). In the latter study, insulation, humidity, temperature, precipitation intensity, support and wind movement were listed as factors that could have contributed to the differences in production. In contrast, exclusion cages reduced herbage yields by ~15% and seed yields by 41-65% of creeping red fescue (*Festuca rubra* L.), where light was reduced and suggested as causative factor (Dobb and Elliott 1964). Studies by Owensby (1969) and others have compared growth between exclusion cages and un-grazed neighbouring pasture.

The empirical data generated from my studies enabled the comparative analysis of pasture growth rates in fenced and unfenced areas under cages Prima facie, given that all pasture plots were protected from grazing, one may expect a similar pasture growth rate and total biomass in both main plots if all other environmental factors are equal. The results were consistent with the hypothesis that exclusion cages can have a positive influence on pasture growth, over and above simply protecting pasture from grazing wildlife. The positive effect could be due to the pasture within the cages having increased access to nutrients and moisture due to the grazing impact on the pastures immediately outside of the cages. However, it is also possible that other environmental conditions that were not measured varied across the caged plots and differentially affected pasture growth. I have suggested that this topic warrants further investigation (Chapter 8).

6.5 Conclusions

This chapter had three aims. The first aim was to quantify the effects of wildlife grazing on the accumulated biomass of four pastures types used for establishing pastures in the study region. My research highlighted some of the difficulties that land owners face when deciding to improve pastures by re-sowing. The research demonstrated that wildlife grazing can severely reduce accumulated biomass in establishing pastures particularly when the area of pasture sown is small. The second aim of the chapter was to test for correlations between wildlife grazing and the botanical composition of improved pasture species, and the amount of bare ground within a pasture. Correlations were established between wildlife grazing and botanical composition and bare ground in improved pastures. Subterranean clover and possibly other legumes may be particularly vulnerable during establishment in areas heavily grazed by wildlife species such as pademelons. Phalaris based pastures appeared adapted to produce higher levels of accumulated biomass in the growing conditions of low rainfall and continuous grazing of the Fosterville property. Heavy grazing during establishment can retard the spread of sown plant cover, providing space for unwanted or less desirable plant species to establish as was demonstrated with ryegrass at Mt. Pleasant.

The final aim of the chapter was to quantify the effects of exclusion cages on accumulated biomass of pastures. My data for Fosterville were consistent with the hypothesis that exclusion cages may influence the microclimate of the pastures under study and may positively affect growth rates under some circumstances. As a consequence, this may lead to an over-estimation of the pasture lost to wildlife grazing under some circumstances. The influence of exclusion cages did not occur at Mt.

Pleasant. Further research into the effects of this design of exclusion cage on pasture growth is required.

Overall, the effect of wildlife grazing on individual sowing events may vary depending on factors such as the wildlife present in an area, the access of grazing wildlife to pastures, the extent of sown pastures, the pasture species used for sowing, and the degree of wildlife control employed. Sowing small areas to pasture may induce plant growth characteristics typical of a grazing lawn as wildlife seek to exploit the relatively higher feed quality of improved pasture species. Given the significant expense associated with improving pastures, controlling wildlife may be a cost effective means of ensuring the longevity of pastures and ultimately the profitability of the pastoral livestock enterprise. This issue is the subject of the research presented in Chapter 7.

Chapter 7: Effects of removing and introducing wildlife control through fencing on production of a newly established pasture

7.1 Introduction

In Chapter 6, I demonstrated that the effects of wildlife grazing on pasture production during pasture establishment can be significant. This research highlighted the potential importance of using wildlife control measures during pasture establishment and the early period following establishment to reduce pasture yield losses to wildlife grazing. The use of poisons such as 1080 to control wildlife in Tasmania's agricultural landscapes and systems is increasingly restricted and is intended to be banned completely by 2016 (see Chapter 1), whilst greater restrictions have been placed on wildlife shooting as a wildlife control measure. Statham and Statham (2010) recommended the use of wildlife proof fencing as an alternative and effective means to control wildlife grazing. They argued that higher pasture yields for livestock grazing can be achieved if the fencing is appropriately constructed and maintained. However, the adoption of wildlife proof fencing as a control measure on extensive grazing properties has generally been limited in Tasmania compared to more intensive systems (Statham and Rayner 1995; Statham and Statham 2010). A number of factors may account for this situation. For example, the productivity and economic value of extensive grazing pastures is often not at the level of that under more intensive grazing systems such as dairying or intensive beef production (Gatenby 2010) and the perceived incentive to invest in wildlife proof fencing may be limited given the 'up-front' costs associated with fence construction and maintenance (Norton *et al.* 2010; Statham and Statham 2010).

The purpose of Chapter 7 is to examine the effects of removing and introducing wildlife control through the use of fencing on the production of newly-established pasture. In doing so, the results of the study can be used to help assess the potential economic benefits of deploying fencing as a wildlife control measure in situations where grazing by wildlife is known or considered to be relatively high. Specifically, the aims of the chapter were to:

- Quantify the effects on accumulated pasture biomass of removing wildlife controls on pasture grazing one year after the pasture was sowed; and

- Quantify the effects on accumulated pasture biomass of introducing wildlife controls on pasture grazing one year after the pasture was sowed.

7.2 Methods

One site at Fosterville and one ex-situ site at Mt Pleasant were chosen to evaluate the impacts of wildlife grazing on pasture establishment. The experimental study was undertaken in two distinct stages. Stage one examined the effects of wildlife grazing on pastures during establishment, and was discussed in Chapter 6. Stage two examined the effects of discontinuing wildlife control and introducing wildlife control one year after sowing of pasture, and is reported here.

7.2.1 Fosterville site

The Fosterville site was located within a degraded pasture and 100 m from the pasture - native vegetation interface (see Figure 2.2 and Chapter 6). The experimental site was established by firstly spraying an application of Roundup® 460CT (a.i glyphosate 460 g/L) at a rate of 3 L/ha with Spraymate® Activator® surfactant (a.i 900 g/L non-ionic surfactant) at a rate of 100 mL/100 L in August 2008 to eradicate the existing pasture. All plots were direct drilled with an Orjyord precision cone seeder on 17th August 2008 at the seeding rates given in Chapter 6, Table 6.3. Irrigation was applied in three applications of 5 mm within a two week period to encourage germination and seedling vigour. The experimental site was surrounded with a 6 strand plain wire sheep exclusion fence allowing access of wildlife to the area while excluding grazing by sheep. Agritone®750 (a.i MCPA present as dimethylamine salt 750 g/L) at a rate of 1 L/ha with Spraymate® Activator® surfactant (a.i 900 g/L non-ionic surfactant) at a rate of 100 mL/100 L was used to control broadleaf weeds in December 2008 and June 2009. Larger thistles were also removed by hand in January 2009. Fertiliser (NPKS 0-6-17-7) was applied in June 2009 at a rate of 140 kg/ha.

A split plot experiment in a randomised complete block design consisting of three blocks was implemented to assess the effect of wildlife grazing on different pastures. There were two main plot treatments; exposed to wildlife grazing, and exclusion of wildlife grazing during establishment. Each main plot was 6 m x 15 m and exclusion of grazing wildlife was achieved using 1.8 m high fencing. The sub-plot treatments consisted of four different pasture types based on cocksfoot, ryegrass, phalaris and a mixture of grasses (see Chapter 6, Table 6.3). The sub-plots were 1.5 m x 15 m in size. Within each sub-plot, four plots were marked for pasture measurements. The experiment was positioned in the centre of the 1.0 ha experimental area with the remaining area sown with the 'Mix' seed combination.

Daytime and spotlight observations, faecal pellets, tracks and infrared digital scouting cameras were used to identify species that were grazing and browsing on the experimental site. It was difficult to estimate the population density of wildlife species, but an indication of wildlife numbers was recorded. As many as 20 wallaby were observed grazing at night on the experimental site and up to 10 kangaroos were commonly observed feeding during late afternoon. Deer were seen in twos and threes, and rabbit and possum were present. The number of animals grazing within the experimental areas varied during the study, but grazing pressure was likely to have been consistently high during this period.

7.2.2 Mt Pleasant site

The *ex situ* experimental site was established by firstly spraying an application of Roundup® 460CT (a.i glyphosate 460 g/L) at a rate of 3 L/ha in May 2008 to eradicate the existing pasture. The area was left in a fallow for four months before being shallowly cultivated to remove the thick mat of organic matter remaining. The organic matter was raked off before a further application of Roundup® 460CT in October prior to direct drilling on the 24th October 2008. Agritone®750 (a.i MCPA present as dimethylamine salt 750 g/L) was applied in early December 2008 at a rate of 0.5 L/ha, and again in late December 2008 (1 L/ha). Further applications of Agritone®750 were applied in March 2009 (0.5 L/ha) and in June 2009 (1 L/ha) to control broadleaf weeds. Irrigation was applied in three applications of 5mm over a two week period in October/November to encourage germination and seedling vigour. Fertiliser (NPKS 0-6-17-7) was applied in June 2009 at a rate of 100 kg/ha.

The Mt. Pleasant site followed the same treatments and split plot RCBD experimental design as that used at the Fosterville site. Sowing rates were increased for the Mt. Pleasant sowing in an attempt to ensure sufficient numbers of plants germinated in the increasingly poor establishment conditions created by lack of rain (see Chapter 6, Table 6.4).

The experimental site (Figure 7.1) was located within a 0.85 ha enclosure containing an established breeding population of wallabies and pademelons. Four wallabies and 15 pademelons were recorded in a count of the Mt. Pleasant wallaby compound in spring 2008. Further counts in 2009 confirmed that these numbers remained steady throughout the study term. Wildlife freely accessed the plots and were observed grazing experimental plots during the day and night.

Figure 7.1 The Mt. Pleasant pasture establishment site in early 2010 following the removal and addition of fences to certain blocks. Note the rank growth of the surrounding pasture.

7.2.3 Removal and introduction of fencing at sites

In November 2009 the fences at the Mt. Pleasant experimental site were rearranged so that only half of the previously fenced plots remained fenced. In addition, half of the previously unfenced plots were fenced. In December 2009 the fences at Fosterville were rearranged so that only half of the previously fenced plots remained fenced. In addition, half of the previously unfenced plots were fenced.

For the purposes of defining treatments and analysing and presenting the results, the initial fencing treatment period which ran from sowing up until the fences were rearranged is termed ‘fenced 2009’ or ‘unfenced 2009’. For the period after the rearrangement of the fencing, the terms ‘fenced 2010’ and ‘unfenced 2010’ are used as most of the period of the treatment was applied in those years.

The design of both sites is illustrated in Figure 7.3 and Figure 7.4, and shown in Appendix 3.

7.2.4 Pasture biomass sampling

Only DM yield data collected from exposed (uncaged) plots were used for the analyses presented below. Measurement of pasture biomass accumulation followed the identical technique as described in Chapter 6. That is, pasture growth and biomass accumulation were measured by periodically harvesting the plots to a height of 40 mm to mimic grazing when the majority of plots in the fenced areas had reached 100-200 mm in height. Plots were cut with manual or mechanical hand shears and a consistent cutting height was achieved by using the top of the quadrat as a height guide. Samples were placed in plastic bags, tied and stored in a food cooler in the field to reduce respiration and water loss. Samples were oven dried at 60°C for 48 hours and weighed to give a DM yield. Following harvest, the plots were mowed to a height of 40 mm with a rotary mower.

7.2.5 Data analysis

To assess the effects of wildlife grazing on the yield of the four pasture types, data from fenced and unfenced plots nested within each pasture sub-plot were pooled and fitted to a linear mixed model using SAS version 9.1 statistical package (SAS Institute Inc. 2004) with block added as a random term, and fence and pasture type and fence by pasture type terms treated as fixed effects. Fence by block was the random term used to test the fence effect, while the other fixed effects were tested with the residual error. This test was performed for each sample date. Fisher’s LSD Post Hoc test was used to compare treatment means.

7.3 Results

7.3.1 Fosterville

The following results section summarises the DM yield grown between 7th December 2009 and 15th April 2010 in altered main plots at Fosterville. Only data from exposed (uncaged) plots were used for these analyses. Data from the four exposed plots nested within the pasture type sub-plots were pooled and averaged.

Pasture biomass for this period was found to be significantly ($P<0.05$) affected by the 2009 and 2010 fencing treatments (Table 7.1). The mean biomass of plots that were fenced in 2010 was 1,529 kg DM/ha compared with 269 kg DM/ha in unfenced 2010 plots (Figure 7.2). This equated to a mean growth rate over the experimental period (Dec-Apr) of 11.9 kg DM/ha/day and 2.1 kg DM/ha/day, respectively.

There was a significant ($P<0.05$) fencing (2009*2010) interaction (Table 7.1). Mean biomass of main plots that were unfenced in 2009, but fenced in 2010 was 1,822 kg DM/ha compared with 1,255 kg DM/ha in main plots that were fenced in both 2009 and 2010. Only in the cocksfoot pasture type was the biomass greater in the fenced 2009 and 2010 main plots than in unfenced 2009/fenced 2010 main plots. There was a significant ($P<0.05$) fencing (2010) by pasture type interaction (Table 7.1). The biomass of cocksfoot appeared most affected by the fencing treatment. The cocksfoot unfenced 2010 treatments produced a mean biomass of 40.1 kg DM/ha which was only 4% of the biomass produced with cocksfoot fenced 2010 treatments. In comparison, the proportion of the biomass grown in other pasture types by unfenced 2010 treatments exceeded 20% of their corresponding subplot by fenced 2010 treatments. Although not significant at $P<0.05$, there was a weakly significant ($P = 0.073$) pasture type treatment effect on pasture biomass. Mean biomass of cocksfoot plots was 1,150 kg DM/ha, compared with 1,028 kg DM/ha for mix, 775 kg DM/ha phalaris and 643 kg DM/ha for ryegrass. There was a reduction in DM yield following the removal of wildlife control in 2010 in all pasture type sub-plots.

Table 7.1 Effect of fencing treatments on DM yield of pasture types at Fosterville

Dependent variable	**num. d.f.**	**den. d.f.**	**F value**	**sig.**
Fencing treatment (2009)	1	6	42.831	.001
Fencing treatment (2010)	1	6	402.624	.000
Fencing interaction (2009*2010)	1	6	7.865	.031
Pasture type	3	24	2.635	.073
Fencing (2009)*Pasture type	3	24	1.498	.240
Fencing (2010)*Pasture type interaction	3	24	4.430	.013
Fencing (2009)*Fencing (2010)*Pasture type interaction	3	24	0.602	.620

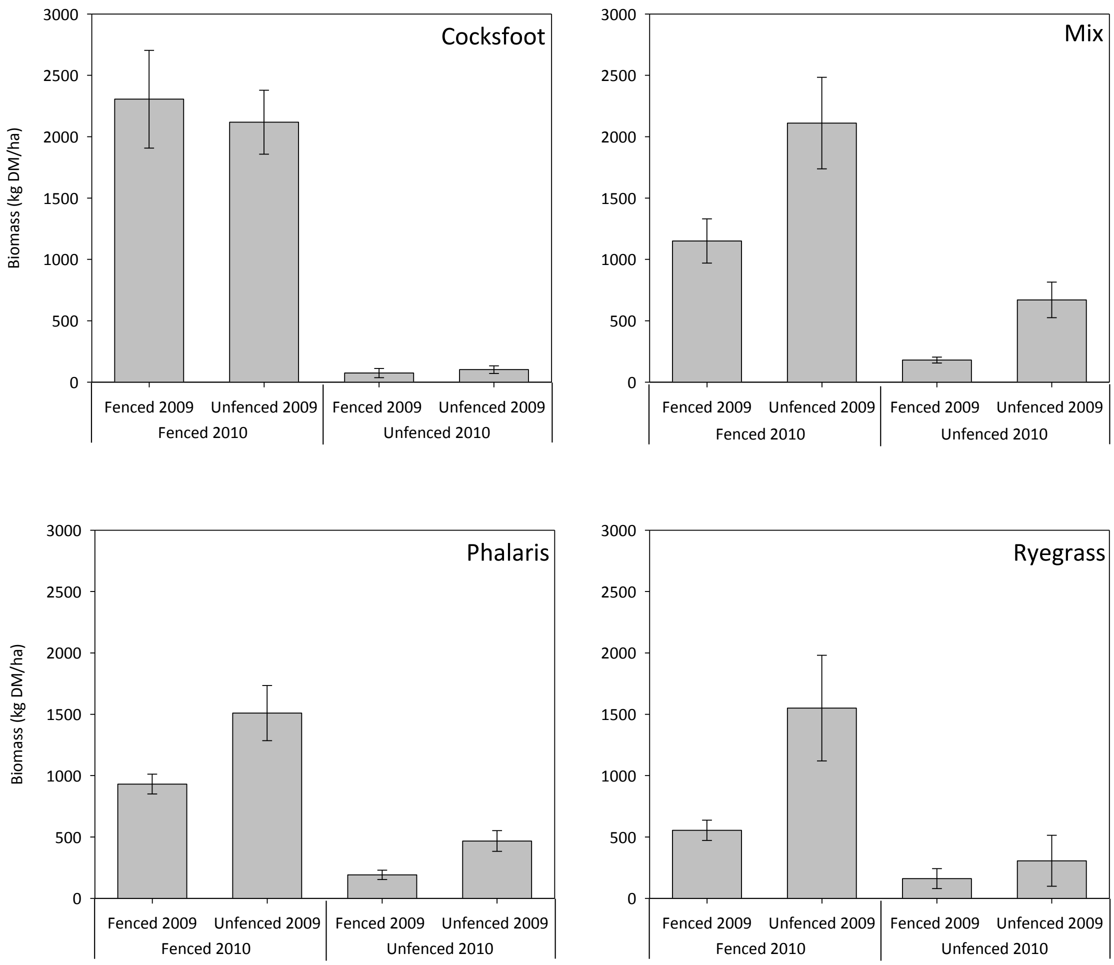


Figure 7.2 Mean DM yield of each of the four pastures types for the period 7th December 2009 – 15th April 2010 at Fosterville. There were four main plot treatments; 1. fenced 2009/fenced 2010, 2. unfenced 2009/fenced 2010, 3. fenced 2009/unfenced 2010, and 4. unfenced 2009/unfenced 2010.

7.3.2 Mt. Pleasant

The following results section summarises the DM yield grown between 10th November 2009 and 23rd March 2010 in altered main plots at Mt. Pleasant. Only data from exposed (uncaged) plots were used for these analyses. Data from the four exposed plots nested within the pasture type sub-plots were pooled and the mean was used in the analysis.

Similar to Fosterville, pasture biomass was found to be significantly ($P<0.05$) affected by the 2009 and 2010 fencing treatments (Table 7.2). The mean biomass of plots that were fenced in 2010 was 893 kg DM/ha compared with 41 kg DM/ha in unfenced 2010 plots (Figure 7.3, Figure 7.6). This equated to a mean growth rate over the experimental period (Nov-Mar) of 6.7 kg DM/ha/day and 0.3 kg DM/ha/day respectively. There was a significant ($P<0.05$) fencing (2009*2010) interaction (Table 7.2). In contrast to Fosterville, mean biomass was greater in main plots that were fenced in 2009 and 2010 (1,168 kg DM/ha) than in main plots that were unfenced in 2009 but fenced in 2010 (619 kg DM/ha). Only in the phalaris pasture type was the biomass greater in the unfenced 2009/fenced 2010 main plots than the fenced 2009 and 2010 main plots.

In contrast to Fosterville, there was no significant ($P>0.05$) fencing (2010) by pasture treatment interaction (Table 7.2). There was also no significant ($P>0.05$) pasture type treatment effect. All pasture types appeared to be affected by wildlife grazing. Implementing wildlife control in 2010 had a positive influence on DM yields with all unfenced 2009/fenced 2010 treatments recording significantly higher yields than unfenced 2009/unfenced 2010 treatments (See Figure 7.4c & 7.4d). In addition, withdrawing wildlife control (fenced 2009/unfenced 2010) caused severe reductions in DM yield for this period (See Figures 7.5a, 7.5b, and 7.6).\

Table 7.2 Effect of fencing treatments on DM yield of pasture types at Mt. Pleasant.

Dependent variable	num. d.f.	den. d.f.	F value	sig.
Fencing treatment (2009)	1	6	8.272	.028
Fencing (2010)	1	6	82.736	.000
Fencing interaction (2009*2010)	1	6	8.923	.024
Pasture type	3	24	1.859	.164
Fencing (2009)*Pasture type interaction	3	24	1.540	.230
Fencing (2010)*Pasture type interaction	3	24	1.725	.189
Fencing (2009)*Fencing (2010)*Pasture type interaction	3	24	2.103	.126

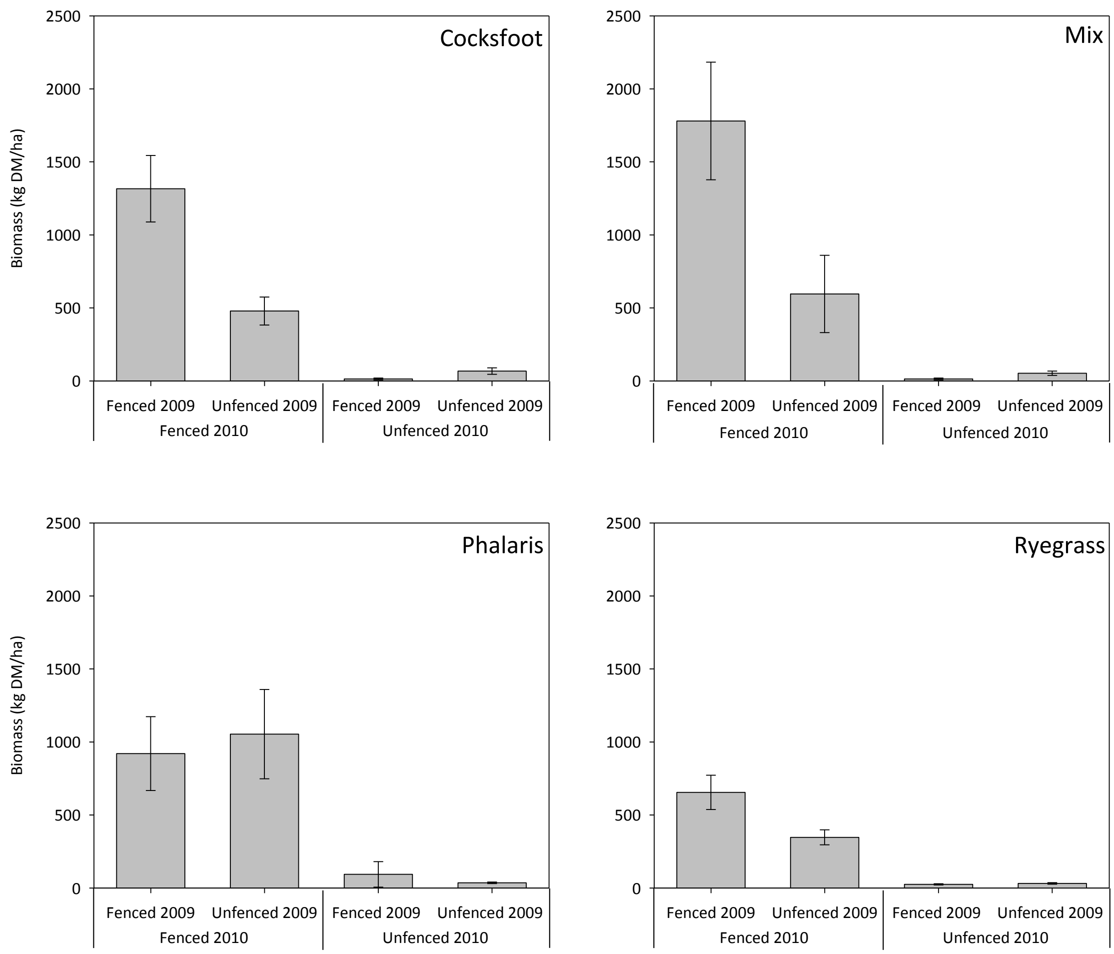


Figure 7.3 Mean DM yield of each of the four pastures types for the period 10th November 2009 – 23rd March 2010 at Mt. Pleasant. There were four main plot treatments; 1. fenced 2009/fenced 2010, 2. unfenced 2009/fenced 2010, 3. fenced 2009/unfenced 2010, and 4. unfenced 2009/unfenced 2010. Error bars represent the standard error of the mean.

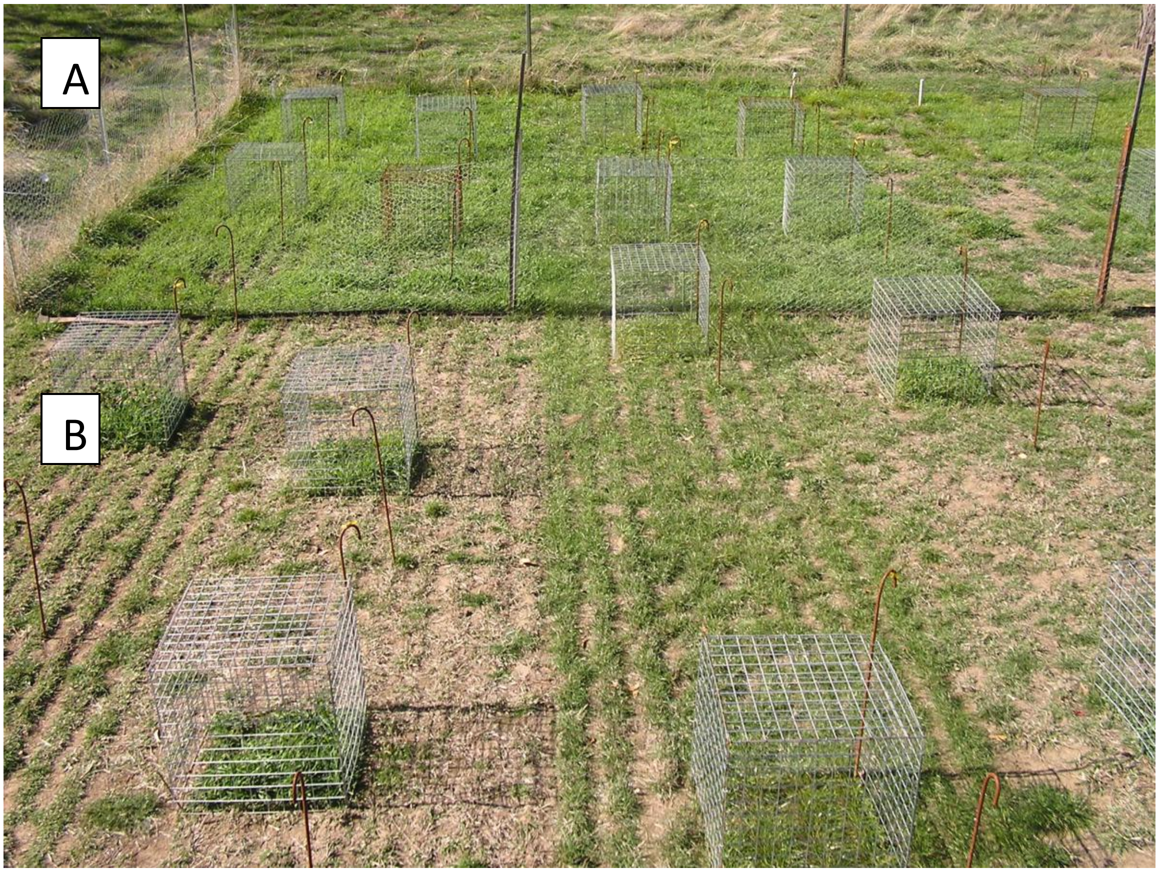


Figure 7.4 This photo shows one of the fencing treatment blocks at Mt. Pleasant. Both the upper and lower areas (A&B) were fenced in 2009. In 2010 the fencing was removed from B, while A continued to be fenced from wildlife. The four pasture types running from left to right are cocksfoot, phalaris, ryegrass and mix.

Figure 7.5 This photo shows one of the fencing treatment blocks at Mt. Pleasant. Both the upper and lower areas (C&D) were unfenced in 2009. In 2010 the fencing was added to C, while D continued to be unfenced from wildlife. The four pasture types running from left to right are cocksfoot, ryegrass, phalaris and mix.

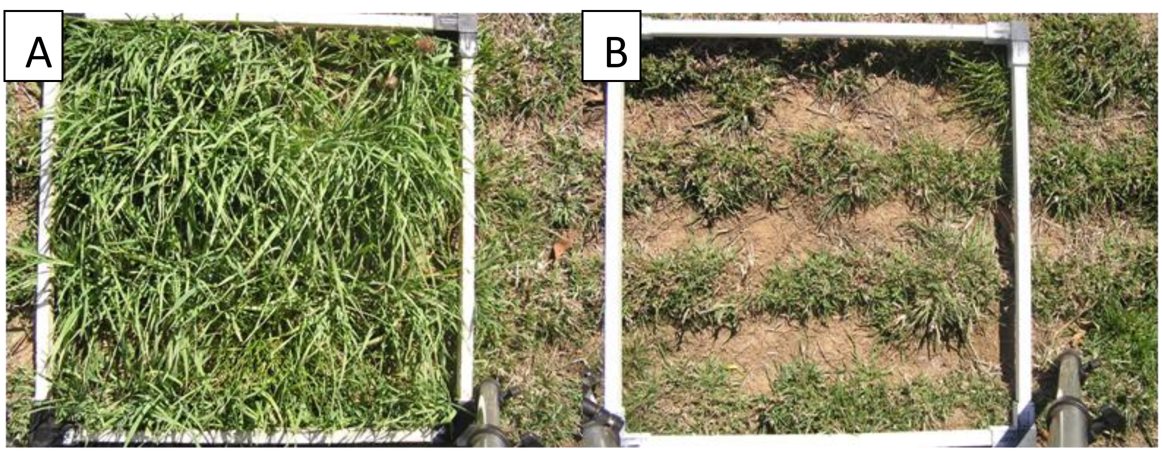


Figure 7.6 This photo shows a plot that was fenced in 2009 and 2010 (A) and another that was fenced in 2009, but had the fence removed in 2010. It illustrates the effect of removing the wildlife control after the initial establishment phase on the DM accumulation of cocksfoot.

7.4 Discussion

Withdrawing wildlife control in 2010 reduced DM yields in all pasture types at Fosterville and Mt. Pleasant. Even ryegrass, previously discussed as possibly less favoured by wildlife at Mt. Pleasant, (see Chapter 6), was heavily grazed. This observation indicated that pastures which are maintained at lower standing biomass by regular grazing (harvests in this case) may be particularly attractive to wildlife given that pastures are senescing over the summer period (December-January) (See Figure 7.1). In addition, the absence of sheep in the Mt. Pleasant system meant that areas not grazed by wildlife quickly became rank, and the experimental areas became one of the few areas with short green grass cover.

At Mt. Pleasant, the recovery of pastures in terms of DM yield was not to the level of pasture that had been fenced in 2009 and 2010. This result was one that could have been expected. In contrast at Fosterville, phalaris, mix and ryegrass pasture types yielded higher DM in unfenced 2009/fenced 2010 than in fenced 2009/fenced 2010 treatments. The ability of pastures to recover from wildlife grazing during establishment may be related to the composition of pasture species. At Mt. Pleasant, unfenced pasture cover in all pasture types was dominated by ryegrass in November 2009. It is unclear whether these pastures would continue to be dominated by ryegrass or change to be dominated by the main grass sown in each pasture type in the absence of wildlife grazing. The concept of predator mediated co-existence (Caswell 1978) and studies by Neave and Tanton (1989) would suggest that greater species diversity within the pasture could be favoured by grazing and that there may be less likelihood of one species dominating. This did not appear to be the case in my study. Rather, my results may indicate that perennial ryegrass is competitive in this grazed environment compared to cocksfoot or phalaris.

The apparent recovery of pastures at Fosterville may be explained by the physiological state of the pastures. Treatments that were fenced 2009/fenced 2010 were observed to have reached the reproductive cycle, and therefore expended some energy reserves in reproduction. Treatments that had been heavily grazed during 2009 (unfenced 2009/fenced 2010) had a chance to recover and enter a reproductive state later, leading to higher yields. It is likely that compensatory growth in these treatments also contributed to the higher yield. There may also have been fewer plants remaining in exposed plots. These plants could have had less competition and thus were more productive once they were protected.

Pastures at Fosterville are likely to respond differently than at Mt. Pleasant. Unlike Mt. Pleasant, at Fosterville both fenced and unfenced treatments were dominated in cover by the main grass species sown. In fact the cover of the main grass sown in each pasture was higher in December in unfenced than fenced treatments except cocksfoot. This would indicate that there was sufficient cover of the sown grasses and that production, over time, could match that of the treatments that were fenced in both years of this study.

Wildlife are able to maintain grazing lawns by regularly grazing pastures to a low standing biomass which restricts the plants ability to enter a reproductive phase (Leonard 2009). Plants growing in such lawns continue to produce new shoots providing there is sufficient soil moisture and suitable temperatures. Grazing lawns typically evolve to contain plants with reduced canopy height, prostate growing habit, and dense canopy (McNaughton 1984). Animals grazing on such lawns can obtain greater grazing efficiency by consuming greater biomass per bite (McNaughton 1984). Grazing lawns are typically more productive per unit area, DM has higher nitrogen content and higher digestibility (McNaughton 1984) than lightly grazed pastures, which may also lead to greater grazing efficiency. The evolution of dentition in the larger macropods to specifically harvest grasses has also improved the efficiency of grazing and early stages of digestion (Sanson 1978; Sanson 1989) and is best suited to short swards. It therefore appears likely that the unfenced pastures, particularly at Mt. Pleasant, would continue to have low levels of accumulated biomass. This highlights the importance of implementing on-going wildlife control measures such as fencing to maximise the standing biomass of pastures for sheep grazing.

Statham and Rayner (1995) used trials in four areas of Tasmania to quantify the loss of pasture production for livestock to grazing wildlife. This research included an examination of the effectiveness of wallaby proof fencing to reduce economic losses from wildlife grazing. At each site, pasture production within 5 ha plots with wallaby proof fencing and with normal fencing were measured and compared. Measurements were made of the number of grazing days from each plot. The trials indicated that properly-installed and maintained wallaby proof fencing could be effective and when grazing wildlife were excluded over 35% more livestock could be supported by the observed pasture production (Statham and Rayner 1995). Norton *et al*. (2010) reported on the effectiveness of wallaby proof fencing for reducing pasture loss to wildlife grazing on beef production properties on King Island, Tasmania. Pasture losses were markedly reduced at sites with fencing and other control measures compared to those with grazing wildlife has free access.

My research has shown that grazing wildlife may have a significant impact on establishing pastures as well as those systems that are already well-established. The results from the wildlife grazing trials at Fosterville and Mt Pleasant indicate that controlling wildlife grazing impacts is important for the success of pasture establishment and appears likely to be significant to ensure that newly-established pastures will both achieve and maintain a relatively high level of productivity over time.

Community perceptions of wildlife management in agriculture continue to evolve and influence options for wildlife control. Wildlife proof fencing has been recommended as an alternative and effective means to control wildlife grazing in many farming situations, but the adoption of this approach remains generally limited in extensive agriculture in Tasmania compared to more intensive systems. Improved understanding of the agricultural and economic significance of pasture loss to wildlife grazing may help to redress the apparent perception in the agricultural community that the ‘up-front’ costs associated with the deployment of wallaby proof fencing as a control measure are prohibitive compared to the possible benefits that can arise from the use of this control measure.

Internationally, fences have been used to mitigate the effects of crop raiding by a number of species. Electric fencing reduced crop raiding by elephants in Amboseli, Kenya, however the effectiveness was related maintenance of the fence and to landscape factors including distance to cover (Kioko *et al.* 2008). Fencing has proved less successful for preventing damage to crops from wild boars *Sus scrofa* in Europe (Geisser and Heinz-Ulrich 2004), but resulted in less intrusions and less damage in studies by Reidy *et al.* (2008) in Texas. A novel method of fencing is being evaluated in Kenya indicating that fencing need not always be made out of timber and wire. Researchers have used the elephant's apprehensiveness to bees to erect lines of bee hives and prevent elephants from entering small farms to raid crops, leading to socioeconomic benefits (King *et al.* 2009).

Proximity to forest edge was also a determining factor in the level of damage by baboons on crops such as maize and cassava in Uganda (Hill 1999). The extent of damage of any pest animal is related to the abundance of the pest, the availability of the valued product, and landscape factors (Hone 2007). Losses in pasture biomass are related to the level of grazing by wildlife and the population density of wildlife. These were classed as damage response determinants by Hone (2007). In the case of my research, the extent of pasture losses is likely to be related to the number and composition of grazing wildlife species, the quantity and quality of the available pasture and the surrounding landscape, with particular reference to distance from the native vegetation.

7.5 Conclusions

This chapter aimed to (i) quantify the effects on accumulated pasture biomass of removing wildlife controls on pasture grazing one year after the pasture was sowed; and (ii) quantify the effects on accumulated pasture biomass of introducing wildlife controls on pasture grazing one year after the pasture was sowed. An experimental (*in situ*) site at Fosterville and *ex situ* site at Mt. Pleasant were used to examine the effects of removing and introducing wildlife control through the use of fencing on the production of newly-established pasture. The study demonstrated that the effects of wildlife grazing on pasture production during pasture establishment, and during the early period following establishment, can be significant. Heavy grazing by wildlife during pasture establishment can lead to initially higher levels of bare ground providing space for less desirable plant species to colonise. I quantified the effects of introducing and removing fencing as a control for wildlife grazing on establishing pastures and demonstrated that the use of fencing as a control can markedly improve the success and longer term productivity of the pasture being established. By introducing wildlife control (fencing) one year after sowing, considerably greater yields were achieved than in plots that remained unfenced (exposed to wildlife grazing). This indicated that if wildlife control could not be implemented at sowing, there is still a benefit of imposing wildlife control after establishment. In contrast, removing wildlife control one year after sowing had a significant negative effect on the yield of plots. This indicated that to achieve optimum production, wildlife control should continue to be implemented after establishment.

In Chapter 8, I consider the economics of wildlife control measures in pasture-based production systems with both established and establishing pastures in the context of the results presented earlier in the thesis.

Chapter 8: General Discussion

8.1 Introduction

Human/wildlife conflicts may arise where human land uses and land use practices detrimentally change the habitat and/or population dynamics of wildlife species. Modification of natural landscapes for agriculture has resulted in many reported impacts on biodiversity (Millennium Ecosystem Assessment 2005), and can lead to an over-abundance of species that are be favoured by agricultural development and the expansion of agricultural land uses. Whether it be elk in North America (Lacey *et al.* 1993), elephant in Africa (Hoare 1999) or primates in Asia (Rao *et al.* 2002), changes in the availability of wildlife habitat and the relative abundance of wildlife species can lead to conflicts in land and resource use. Within Tasmania, one of the factors affecting production in the pasture-based agricultural industries is grazing by native wildlife, in particular pademelon, wallaby and possum (Statham and Rayner 1995) that are now over abundant partly due to agricultural development (Driessen and Hocking 1992). The 'Alternatives to the use of 1080' program was established in Tasmania to develop science-based alternatives to the use of 1080 poison for the control of wildlife that graze pastures.

My thesis reports the results of a two and a half year study consisting of a range of experiments at Fosterville, a sheep grazing property in the Midlands region of Tasmania, and *ex-situ* experiments at the TIA's Mt. Pleasant Research Laboratories site in Launceston. I investigated the nature of wildlife grazing on established pastures with distance from native vegetation and across seasons, and the influence of wildlife grazing on pasture species composition, ground cover in pastures, pasture establishment and soil health.

The cleared and modified land at Fosterville is typical of many parts of the Midlands region, consisting of a combination of improved, semi-improved, and native pastures (Kirkpatrick and Bridle 2007). Fosterville supports merino wool production as well as meat production in the form of prime lambs, mutton and beef (S. Foster pers. comm.. 2007). The wildlife observed on the property included native species such as kangaroo, wallaby, pademelon, possum and wombat, and introduced species such as deer and rabbit. Quantitative (non-parametric) population models were developed for wallaby, kangaroo, possum, and rabbit using the wildlife survey count data (Chapter 2, Section 2.11). The number of sightings of pademelon and deer were insufficient for statistical analysis. Analysis of the count data suggested that animal numbers reached a maximum within the 0-300 m region from native vegetation in the area sampled (Chapter 2, Section 2.11). The kangaroo, wallaby and possum were relatively abundant, and typically associated with most of the observed pasture losses resulting from wildlife grazing. Limitations associated with the use of different techniques to survey and monitor vertebrate species such as macropods were discussed earlier in the thesis (Chapter 2, Section 2.11). Overall, however, due to various constraints it was not possible to establish the precise contributions of each wildlife species to pasture loss. Rather, pasture loss and the influence of wildlife grazing on pasture and soil factors were considered in the context of the observed relative abundance and foraging behaviour of the different wildlife species in both space and time.

As discussed earlier in the thesis (Chapter 2, Section 2.5), the native vegetation communities of the Midlands region of Tasmania have been cleared and significantly modified for agriculture (Kirkpatrick *et al.* 2007). As a consequence of the major changes that have occurred in ecosystem processes, it was not possible to identify and replicate true control sites to study the possible impacts of wildlife grazing. Instead, my *in-situ* experiments had to be based on the pseudo-replication of control sites and impact sites (Hurlbert 1984; Stewart-Oaten *et al.* 1986) to examine changes in environmental conditions in pasture systems that could be attributed to wildlife grazing. The limitations of this approach were considered earlier in the thesis (Chapter 2, Section 2.13). To help redress these limitations, I employed a scientific framework to establish a time series of data on variation in wildlife grazing at Fosterville and made maximised the replication of the field research and the length of monitoring and sampling as much as possible (see

Hurlbert 1984). The results of the field studies were interpreted with caution and, where possible, further tested and evaluated using *ex situ* experimentation at Mt Pleasant.

8.2 Key research findings

Wildlife grazing and pasture production

Grazing of pastures by wildlife at Fosterville resulted in significant reductions in pasture production. However, similar to other studies in Tasmania including Statham and Rayner (1995), Statham (2000), Donaghy and Tegg (2001) and Norton *et al*. (2010) and elsewhere in Australia including studies by Hill (1988), Arnold (1989) and Barnes and Hill (1992) losses were variable. For example, over 90% of pasture production within the first three hundred metres of native vegetation was consumed by wildlife in late summer and autumn during the study. In comparison, pasture production loss to wildlife grazing was significantly lower or negligible, depending on the season, at a distance of 800 m from the edge of native vegetation. That is, the amount of pasture lost to grazing was inversely related to distance from the edge of native vegetation. Changes in the relative abundance and foraging behaviour of wildlife species were consistent with the observed loss in pasture production. For example, both the weight of faecal pellets and the percentage reduction in pasture production decreased with distance from the edge of native vegetation. Similar observations were reported by Southwell (1987), While and McArthur (2006), and Norton *et al*. (2010) while examining population dynamics, foraging patterns and habitat use by macropods. Grazing behaviour of wildlife at Fosterville compared well with recognised predator avoidance behaviour studied by Banks (2001). The trade off between obtaining feed and increased risk of predation was observed. Animals grazed mostly at night, close to cover, and when disturbed flushed into the native vegetation.

Seasonal variation in pasture production (the amount of herbage available to the wildlife, affected the proportional loss of pastures to grazing over time. The measured proportion of pasture loss in my study was greatest during summer, autumn and winter harvests and least during spring. Pasture growth rates were greatest during spring, indicating that feed availability was likely to influence the proportion of pasture loss. Fletcher (2006) noted that the pasture biomass eaten increased with feed availability, however, the relative greater availability of pasture during spring in this study would mean a reduction in the proportion of pasture being lost. Fewer faecal pellets were found during spring, and faecal pellets have been shown to be positively related to the amount of feed eaten (While and McArthur 2006). Lower numbers of wildlife feeding on pasture during spring can be explained by higher herbage availability within native vegetation areas, although this was not measured. With more pasture available during spring, it was suggested that the percentage pasture loss to wildlife grazing would decrease, given that individual animal intakes are unlikely to increase to the extent of keeping up with pasture production. This pattern was observed during the study.

The impact of wildlife grazing on pastures has been reported by a few previous short term studies in Tasmania (e.g. Statham and Rayner 1995, Statham 2000, Donaghy and Tegg 2001). For example, Statham and Rayner (1995) reported variation in the amount of pasture loss to wildlife grazing, recording a loss ranging between 17-100% over 6 study sites, with 5 of the 6 sites recording pasture DM reductions in excess of 40%. Substantial variation in the amount of pasture loss resulting from wildlife grazing was observed during these studies and was attributed to factors such as seasonal variation in pasture production and temporal changes in wildlife movements and foraging behaviour. These findings are consistent with theory discussed by Hone (2007) that the extent of damage of any pest animal may be related to the abundance of the pest, the availability of the valued product, and various landscape factors. In this case, the extent of pasture losses is likely to be related to the number and composition of grazing wildlife species, the quantity and quality of the available pasture, and the surrounding landscape, with particular reference to distance from the native vegetation.

Wildlife grazing and the botanical composition of pastures

The difference in observed and modelled pasture growth rates in enclosed plots indicated that there may have been a number of indirect effects of wildlife grazing on the botanical composition of the pastures and the extent of ground cover. Generally, heavy grazing by wildlife has resulted in reduced height and cover of grasslands (Neave and Tanton 1989;

Fletcher 2006). For example, in the current study, the DM production of cocksfoot, was influenced by wildlife grazing and protection from grazing resulted in a steady increase in the DM production and ground cover of this pasture species. The influence of wildlife grazing on the amount of bare ground was most evident closer to the edge of native vegetation where the highest losses in pasture production were recorded. Wildlife grazing in this area during the period of drought resulted in both pasture loss and the death of individual pasture plants. As rainfall increased during the latter part of the study, pasture plants increased in cover abundance in areas exposed to high levels of wildlife grazing, and the extent of bare ground became similar in plots exposed to grazing and protected from grazing. This is supported by Robertson (1987) who noted that changes to pastures are influenced greatly by seasonal and rainfall events.

Wildlife grazing and soil health

Soil chemical and biological processes can influence the productivity of pastures. It was hypothesised that health of soils in the plots exposed to wildlife grazing may be influenced. For example, browsed plots that have less pasture biomass and less ground cover may have lower soil nutrient levels, less soil microbial activity and poorer root structure than plots protected from grazing. However, no observed differences were found in the levels of ammonium N, nitrate N, total N, organic C, pH, or electrical conductivity of the soil sampled under pastures that were either protected from grazing or exposed to grazing by wildlife. It was also hypothesised that as pasture production was higher in the enclosed treatment plots, then both higher levels of organic C and total microbial biomass would be found in these plots compared to the plots exposed to wildlife grazing. However, no statistical difference in these features were found between the protected and exposed plots at the $P<0.05$ level. Cotching (2009a) noted that detectable changes in soil organic carbon levels may take several years, and this may also be the case for the other variables measured.

The root biomass of pasture species (0-150 mm depth) appeared related to pasture species composition and increased with distance from native vegetation. The highest root biomass was found in areas dominated by phalaris which is a deep-rooting and drought tolerant species. Lower levels of root biomass were found in areas dominated by annual grasses or perennial ryegrass, these pasture species have relatively shallower roots than phalaris (Blair 1997). A further hypothesis was that the protection of pastures from wildlife grazing would have a positive effect on root structure and root biomass. However, there was no measured effect on root biomass found after 2 years of the study. Differences in root biomass attributable to grazing intensity were reported by King and Hutchinson (1976) following a 10 year imposed grazing study, while none were observed by Lodge *et al.* (2006) during a much shorter study in New South Wales. It is possible that the duration of my field studies may not have been long enough for differences in root weight between these grazing treatments to be expressed.

Overall, there was no observed effect of wildlife grazing on the measures of soil health examined in this study. The lack of a significant difference in some of the variables examined may be due to the improved growing conditions found at Fosterville over the last 9 months of the study. Above average rainfall produced a flush of growth which provided feed for wildlife in the woodland areas of the Macquarie Tier, thus reducing the grazing pressure on the experimental area. With an increase in pasture production occurring on all plots, any actual differences between treatments may have been reduced. Future studies of this nature may also consider the influence of factors such as the size of exclusion cages and the duration of grazing intensity experiments on soil health indicators.

Wildlife grazing and pasture establishment

There were a number of important findings arising from the establishment experiments conducted at Fosterville and Mt. Pleasant. Wildlife grazing had a significant effect on pasture biomass accumulation (similar to established pastures) during establishment in treatments exposed to wildlife grazing. Heavy grazing during pasture establishment can lead to initially higher levels of bare ground providing space for less desirable plant species to colonise. Subterranean clover was particularly vulnerable to heavy wildlife grazing during establishment trials at Mt. Pleasant, but not at Fosterville. This difference may indicate that variations in wildlife grazing behaviour and the legume content of pastures surrounding areas where pasture is being established can influence the planting success. Phalaris-based pastures appeared best adapted for establishment under situations where wildlife grazing pressure was high. Controlling wildlife during the pasture establishment phase increased both the likelihood of establishment success, and pasture production at these sites in subsequent seasons. This observation highlighted the need for the use of wildlife controls both during and following the establishment of pastures. A possible positive influence of exclusion cages on pasture growth was identified, although these results were not consistent across both sites. Exclusion cages have been known to have both positive and negative effects on pasture growth (t' Mannetje 2000a). Exclusion cages may influence resources (water, nutrients) available for plants. Further research is required to confirm the influence of exclusion cages.

Conceptual models of grazing and plant-animal interactions in heterogeneous landscapes

Traditional approaches to grazing management and modeling grazing interactions in heterogeneous landscapes often attempt to use a limited number of variables (e.g. wildlife species, number of animals, forage requirements) and assume equilibrium conditions to generate predictions (Walker and Heitschmidt 1986; Holechek *et al.* 1995; Bailey *et al.* 1996; Dumont *et al.* 2002; Briske *et al.* 2008). Some studies have moved beyond assumptions of equilibrium conditions and have considered the effect of vegetation patterns and scale on the feeding patterns and diet selection of animals (Swift 1987; Edwards *et al.* 1994; Clarke *et al.* 1995; Parsons *et al.* 2001; Chapman *et al.* 2007). However, Laca (2009) argued that progress in the understanding and management of grazing systems requires greater consideration of factors such as spatial heterogeneity, event-driven dynamics, and scaling effects in the ecosystems under study. Laca (2009) suggested that novel concepts, and new or hybrid models and tools are required to promote progress in grazing science. In particular, he recommended that modeling and management approaches incorporate heterogeneity and nonlinear scaling of spatially and temporally distributed ecological interactions such as diet selection, defoliation, and plant growth (Laca 2008; Laca 2009).

Veblen and Young (2010) recognized the significance of heterogeneity and nonlinear ecological interactions in their study of the contrasting effects of cattle and wildlife on the vegetation development of a savannah landscape mosaic in Kenya. They reported that through their effects on plant communities, herbivores may have strong direct and indirect effects on savannah ecosystems and have the potential to create and maintain savannah landscape heterogeneity. The development and maintenance of vegetation may be controlled in part by herbivory (Veblen and Young 2010). Their results demonstrated that different combinations of cattle and wildlife have different effects, largely due to contrasting forage preferences, on the persistence of landscape heterogeneity in vegetation. Their work provided some evidence for contrasting effects of cattle and wildlife on short-term plant interactions and successional processes within the herbaceous plant community under study (Veblen and Young 2010).

A number of ecological approaches to modelling plant-animal interactions have been proposed to improve understanding and prediction of ecosystem dynamics and the functional response and resilience of ecosystems to disturbance across scales. These approaches include theoretical and empirically-based models, mechanistic and process-based models including models of functional response, and hydrid and novel models (Holling 1965; Murdoch 1973; Pickett *et al.* 1987; Westoby *et al.* 1989; Roughgarden 1997; Bestelmeyer *et al.* 2004; Bestelmeyer *et al.* 2009). State and Transition Models (STMs) have been relatively commonly discussed in the scientific literature over the past two decades. These models are a conceptual construct that can help identify patterns and mechanisms of ecosystem response to natural and anthropogenic drivers (Bestelmeyer *et al.* 2009). The models are based on the assumption that ecosystems exhibit threshold behaviour in which transitions to alternative ecosystem states occur that are difficult to reverse (Pickett *et al.* 1987; O'Neill *et al.* 1991; Milne *et al.* 1992). If appropriate data are available, an STM approach may provide a detailed understanding of the range of states, transitions, and thresholds possible in an ecosystem, and a summary of processes driving the system (Pickett *et al.* 1987; Parsons and Dumont 2003).

Much of the published literature on these conceptual approaches has focused on the use of such models to better understand ecosystem dynamics and to try to provide stronger links between the research and management community. STMs and related models, for example, have been used to inform management programs for national parks in the USA (Fancy *et al.* 2009) and as a basis for a framework for rangeland management (Westoby *et al.* 1989; Bestelmeyer *et al.* 2004; Briske *et al.* 2005; Bestelmeyer *et al.* 2009). They have been promoted in agricultural landscapes in Australia as a suitable framework for understanding the ecology of complex ecosystems and as a guide to restoration and management activities (Yates and Hobbs 1997; Prober *et al.* 2002; Dorrough *et al.* 2005; Spooner and Allcock 2006; Rumpff *et al.* 2010). A state and transition approach (Vegetation Assets, State and Transitions (VAST)) has been developed for assessing vegetation condition at the Australian national scale including across agricultural landscapes supporting pasture-based primary industries (Thackway and Lesslie 2006; Lesslie *et al.* 2010).

In an Australian regional context, Rumpff *et al*. (2010) developed a STM for vegetation condition for the Goulburn Broken catchment in the context of an Adaptive Management framework. The Namoi Catchment Management Authority in NSW identified the value of STMs as a means to establish thresholds of regeneration capacity for their Catchment Action Plan (Namoi CMA. 2010). A similar modelling approach was used by Grant (2006)to help guide the restoration of Jarrah forest sites mined for bauxite. Based on the success of this modelling, Grant (2006) suggested the approach had the potential to be used in other land management contexts such as forestry and agriculture.

I considered the use of conceptual models such as models of functional response, State and Transition Models and other approaches to evaluate and interpret the empirical data on wildlife grazing and vegetation response collected during the present study. However, the potential utility of these approaches appeared limited given my primary focus on pasture-based grazing impacts. This focus meant that I only examined a portion of the ecosystems routinely exploited by the native herbivores at Fosterville. In the modified landscapes in the Midlands region of Tasmania it is possible for different grazing wildlife species such as macropods and deer to relatively quickly move (each day and seasonally) between native vegetation communities and introduced pasture-based systems to forage and meet their nutritional requirements. In the absence of data on browsing and foraging by wildlife in the range of native vegetation communities found on and around Fosterville it was not feasible to develop reliable models of the grazing interactions and the functional response of herbivores to food availability. Nor was it possible to develop a full understanding of the response of the vegetation to herbivory under the range of conditions operating in the study region.

Some studies of foraging and diet selection by macropods have shown that the diversity of food options, and access to food of variable palatability and nutritional value will almost certainly influence foraging behaviour (Short 1987; Jarman and Phillips 1989). Unfortunately, however, it was not generally feasible to determine grazing impact across all vegetation communities exploited by the grazing wildlife species present at Fosterville. Nonetheless, while my research findings in this thesis focus on understanding the implications of wildlife grazing for pasture-based systems, in Section 8.5 of this Chapter I outline a number of issues concerning the broader management of wildlife impacts on native vegetation communities in fragmented and variegated landscapes such as those found in the Midlands region of Tasmania that warrant research.

One constraint in modeling wildlife grazing dynamics is the lack of accurate and repeatable techniques for efficiently discriminating grazing effects from both temporal variability and spatial heterogeneity of vegetation communities. Both forms of variability can make it difficult to discern grazing system effects on vegetation response and forage production (Weber *et al.* 1998; DeVries *et al.* 1999). Remote sensing may be an efficient tool for detecting differences in spatial and temporal patterns of grazing impact on vegetation, but applications of the technology for this purpose are yet to be fully evaluated in a range of landscapes including those supporting pasture-based grazing systems. Blanco *et al*. (2009) examined the potential of remote sensing data (Normalized Difference of Vegetation Index (NDVI) data from Landsat Thematic Mapper images) to monitor grazing patterns and vegetation change in a semiarid region of Argentina. Although the technique was unable to discriminate between forage and non-forage components of the vegetation, Blanco *et al*. (2009) suggested that the approach had value because the overall impacts of grazing on vegetation could be compared both spatially and temporally. Recent improvements in the spatial resolution of satellite imagery and studies examining the application of remote sensing techniques to monitor changes in vegetation extent and condition (Miura and Jones 2010; Jones *et al.* 2012) may provide a quantitative basis to eventually enable the fine-scale discriminate of grazing effects on vegetation communities in heterogeneous landscapes.

Economics of controlling wildlife grazing

My research indicated that the concerns expressed in the Tasmanian farming community about the impacts of native wildlife on pasture production and lost farm income appear to be well-founded. The measured range in pasture loss to wildlife browsing is close to that first reported by Statham and Raynor (1995). Based on comparable field-based wildlife grazing studies to those undertaken for my research, Norton *et al*. (2010) found that the average pasture loss to species such as Tasmanian Pademelon, within 100 m of the bush-line, was close to 60% of total pasture production at study sites in north west and north east Tasmania. These authors estimated that, on average, 85 Tasmanian Pademelons could be maintained on this level of pasture consumption (kg DM ha^{-1} yr^{-1}). If the average loss of pasture production at these sites was prevented and directed to production, the biomass of pasture saved would be sufficient to support 15 DSE (Norton *et al*. 2010). It is now clear that pasture loss to wildlife grazing can be both significant and geographically-widespread in Tasmania.

The economic impost of wildlife grazing at Fosterville was calculated by multiplying the amount of pasture lost (tonne DM ha^{-1}) by the cost of a tonne of feed. During 2009 the estimated average pasture loss over the first 800 m from the edge of native vegetation was 1830 kg DM ha^{-1} or AUD \$366 ha^{-1}. Another way to approximate the costs of wildlife grazing is to estimate the opportunity cost, due to pasture loss from wildlife grazing, of grazing more sheep. The loss of an estimated 1830 kg DM ha^{-1} equated to the loss of 6.5 DSE ha^{-1}.

Norton *et al*. (2010) used satellite imagery to estimate that around 50% of the total area of farm pastures across Tasmania occurs within 300 m of native vegetation and suggested that many of these pastures could be subject to high levels of browsing by wildlife if the patterns observed during their study hold generally in Tasmania and over time. Based on these observations and my research findings, the economic impost of wildlife grazing on many pasture-based agricultural production systems in Tasmania is likely to be significant.

Overall, the perception that the up-front costs associated with the construction and maintenance of wallaby proof fencing to control wildlife are prohibitive is likely to be incorrect for many properties. The cost of using wallaby proof fencing may be in the order of AUD $2,500 - $3,000 per km, on average, in Tasmania (Statham and Statham 2010). Norton *et al*. (2010) have recommended that a benefit cost analysis be undertaken on properties where wildlife grazing is known to be high to determine the economic returns that are likely to be achieved by installation of wallaby proof fencing. In many instances, my research suggests that the installation of fencing as a control on wildlife grazing could generate significant economic returns over time.

8.3 Findings in relation to the thesis aims

The public controversy over the use of 1080 poison for controlling native wildlife in Tasmania was a significant precursor to a decision by the State Government in 2004 that the use of 1080 would be banned on public land and alternatives to its use had to be found. As a consequence, the “Alternatives to the use of 1080” programme was established and, as part of a broader study, I undertook a research to quantify and monitor the impact of native herbivores on farm pastures, and to provide scientific information for the management of grazing wildlife in the Tasmanian Midlands. My research had six specific aims. These aims were systematically addressed using experimental techniques and *in situ* and *ex situ* study sites. The outcomes of the research are considered below in relation to each aim.

Quantitatively investigate wildlife grazing on established pastures by testing for correlations between pasture loss to grazing and distance from native vegetation, and season.

My research established correlations between pasture loss from wildlife grazing and distance from native vegetation. On average, the greatest loss of pastures to wildlife grazing occurred close to native vegetation and declined with distance from native vegetation. Based on direct and indirect observations, different species of wildlife were involved in grazing of pastures at distance to native vegetation, and these differences were consistent with the known foraging behaviour of the wildlife present during the study period.

Correlations were established between the extent of pasture loss to wildlife grazing and season. Seasonal variation in pasture production was related to the proportional loss of pastures to grazing over time. Measured pasture loss was greatest during summer, autumn and winter harvests and least during spring. Lower numbers of wildlife fed on pasture during spring and I suggested this pattern occurred because of the higher herbage availability to wildlife within native vegetation areas at this time.

Test if a relationship exists between observed grazing damage and an index of feeding activity by native herbivores.

My research demonstrated a relationship between observed grazing damage and an index of feeding activity (based on the weight and number of faecal pellets per unit area). Changes in the relative abundance and foraging behaviour of wildlife species were consistent with the observed loss in pasture production, and the amount of pasture lost to grazing was inversely related to distance from the edge of native vegetation. Both the weight of faecal pellets and the percentage reduction in pasture production decreased with distance from the edge of native vegetation.

Quantify the effects of wildlife grazing on pasture species composition and ground cover in pastures.

I quantified the effects of wildlife grazing on pasture species composition and demonstrated that grazing has direct and indirect effects on both the growth and likely survival of pasture species. The difference in observed and modelled pasture growth rates in enclosed plots indicated that there may have been a number of indirect effects of wildlife grazing on the botanical composition of the pastures and the extent of ground cover. Bare ground was most evident near the edge of native vegetation where grazing was highest.

Test for correlations between wildlife grazing and soil health and root biomass in pastures.

My research did not identify any correlations between wildlife grazing and soil health and root biomass in pastures. It is possible that any such correlations were 'masked' or unable to be detected due to factors such as the relatively limited duration of my field study, and the fact that the pasture growing conditions found at Fosterville over the last 9 months of the study improved due to above average rainfall. The use of relatively small wildlife exclusion cages in my study may have affected the microclimate and, as a consequence, the soil health of the pastures being examined. I have suggested that this aspect may warrant further study.

Quantify the impacts of wildlife grazing on establishing pastures and measure the effects of wildlife controls on pasture production.

My research quantified the impacts of wildlife grazing on pasture establishment. My research demonstrated that wildlife grazing can have a significant effect on pasture biomass accumulation during establishment. Heavy grazing by wildlife during pasture establishment can lead to initially higher levels of bare ground providing space for less desirable plant species to colonise.

I quantified the effects of introducing and removing fencing as a control for wildlife grazing on establishing pastures and demonstrated that the use of fencing as a control can markedly improve the success and longer term productivity of the pasture being established.

Quantify and evaluate the economic costs of wildlife grazing in the Midlands region of Tasmania.

The empirical results of my research enabled a quantification and evaluation of the economic costs of pasture loss to wildlife grazing in the Midlands region. I calculated that the average pasture loss over the first 800 m from the edge of native vegetation at Fosterville during 2009 was valued at AUD \$366 ha^{-1}. I calculated that the pasture lost to wildlife grazing equated to a reduction in livestock carrying-capacity at Fosterville of 6.5 DSE ha^{-1}. Based on these estimates, the loss of pastures to wildlife grazing in comparable areas of the Midlands is likely to be significant. I concluded that in many instances the installation of fencing as a control on wildlife grazing could generate significant economic returns over time.

8.4 Implications for management

A number of findings arising from the experiments outlined in the research chapters of the thesis have management implications and are considered below. These comments have been written with land managers and personnel involved natural resource management and environmental stewardship in mind.

- *Wildlife grazing may significantly reduce accumulated pasture biomass on farms, especially in paddocks close to native vegetation*

Findings from the experiments in Chapter 3 and Chapter 6 investigating the effects of wildlife grazing on pasture production clearly indicated that the losses can be substantial. It was also clear that pasture loss to wildlife grazing may be significantly higher close to the native vegetation interface on farms. Therefore, control of wildlife could help to increase the availability of pastures for livestock both seasonally and during periods of drought.

- *Monitoring of wildlife and wildlife controls may be required on an on-going basis in regions where wildlife populations are relatively high and species may forage widely*

The intensity of wildlife grazing on pastures can vary in space and time depending on factors such as wildlife population levels, food availability, the mobility of different species and the energetic requirements of animals during breeding. Hence, on-going monitoring of wildlife population levels and the loss of pastures to wildlife grazing may be necessary to determine when to implement wildlife controls.

- *Wildlife controls may be required on farms during critical periods such as the establishment of pastures or periods of drought when competition for food between livestock and grazing wildlife may be significant*

Control of grazing wildlife during the establishment phase of a pasture can be critical for successful establishment. The pasture establishment study conducted at Mt. Pleasant showed that wildlife grazing can result in the removal of sown pasture species before they become established, and increase the likelihood of weed invasion when the extent of bare ground increases due to overgrazing. These changes may produce a significant reduction in pasture biomass, reduce the nutritional value of the newly-established pasture, and reduce the extent of legumes at a site able to fix soil N.

- *The process of pasture establishment on farms could be improved to help minimise the potential impacts of wildlife grazing*

The process of pasture establishment on farms with significant populations of grazing wildlife species could include the following steps: (i) matching the pasture species to be established with the expected level of grazing pressure, sowing phalaris for example, (ii) use of wildlife controls during establishment, (iii) following establishment continue to monitor for heavy wildlife grazing, (iv) ensure wildlife controls are in place during periods of feed shortages, (v) concentrate the wildlife control effort on paddocks close to native vegetation, and (vi) consider the installation of suitable wallaby-proof fencing in areas where wallabies are the predominant wildlife species.

- *In low rainfall environments phalaris could be a suitable pasture species where the impact of wildlife grazing on pasture production may be significant*

Phalaris is the major component in many pastures at Fosterville because it is tolerant of dry conditions, water-logging and insect pasture pests, and has good growth from autumn to spring (Johnson *et al.* 2006). As highlighted in Chapter 6, the production of phalaris-dominant pastures was greater than ryegrass and cocksfoot based pasture types under grazing pressure by wildlife. This suggests that phalaris may be more able to withstand heavy and continued grazing by wildlife when establishing in low rainfall areas. Sowing phalaris-based pastures in paddocks close to native vegetation that provides habitat for grazing wildlife may ensure that higher DM production is achieved than would otherwise be possible.

- *Monitoring faecal pellets can help to assess the relative abundance of grazing wildlife and their potential grazing impact on pastures*

My results suggested that counts of wildlife faecal pellets may be useful as an index of feeding activity and provide a quick and inexpensive means of identifying the severity of wildlife grazing. This may avoid the need in the first instance to use exclusion cages and pasture measurements to assess grazing impacts. Instead, the latter methods could be employed simply to re-assess the findings of the wildlife faecal pellet counts before wildlife control measures are deployed.

- *'Wallaby-proof' fencing may be useful for controlling wildlife, but needs to be designed with the local conditions in mind*

The use of exclusion cages in my study promoted the benefits of excluding wildlife from pastures. However, the nature of the fencing and its location on a property need to be designed with the local conditions in mind, including factors such as the grazing wildlife species present, and their use of the immediate and regional environment. The exclusion of grazing wildlife from improved pastures at Fosterville would require the use of a combination of deer and wallaby fencing (see Statham and Statham (2010). This type of fencing could exclude kangaroo, wallaby, pademelon and deer, but not possum or rabbit. Since the former four species are considered to have the most significant grazing impact on pastures, use of this type of fencing should significantly reduce pasture loss. Statham and Statham (2010) discussed issues associated with the installation of wallaby-proof fencing including the need to reduce pressure on the new fence from resident animals seeking access to pastures, and consideration of the movements of non-target wildlife such as the wombat.

- *Shooting programs should be implemented during times of low pasture availability*

Greatest reductions in accumulated biomass were recorded during low pasture availability. The carrying capacity of properties is determined by the number of stock which can be grazed during these times. By reducing the grazing impact of wildlife, particularly during periods of low pasture availability, the carrying capacity of livestock and therefore animal production can be improved. As indicated by the feeding activity, wildlife may spend more time feeding on improved pasture during periods of low pasture availability. Therefore improvements in shooting or poisoning efficiency can be made due to greater numbers of wildlife exposed on improved pastures.

- *Management of grazing wildlife on farms needs to recognise the broader range of issues associated with the over-abundance of wildlife*

Wildlife grazing farm pastures are also reliant on native vegetation for a portion of their sustenance. For example, analyses of the gut contents of macropods shot at Fosterville, in a pilot study (unpublished) not detailed in this thesis, indicated that large portions of the diet of individuals can be attributed to browsing of native plant species. Open woodlands and more heavily forested areas on the Macquarie Tier that are adjacent the pastures at Fosterville are reported to be heavily used as a source of herbage by wildlife species (S. Foster pers. comm. 2009). The installation of wallaby-proof fencing may limit wildlife access to paddocks and reduce pasture losses, but subsequently increase pressure on native vegetation communities. Hence, the sustainable management of over-abundant species of grazing wildlife will require a more holistic approach in these regions if biodiversity conservation and animal welfare issues are to be addressed as well as agricultural production concerns.

The deployment of wildlife controls is not necessarily straight forward. In order to achieve the desired benefits from wildlife control, it is important that populations are reduced to levels at which their impacts are acceptable to the landowner. In some instances the relationship between pest population density and level of impact is non-linear (Hone 2007). In these cases a certain reduction in pest population density will not always provide a proportionate reduction in impact. For example, Norton and Johannsohn (2010) found a positive non-linear relationship between the abundance of wallaby and pasture loss. In this case reducing the number of wallabies by half would only result in a decrease in pasture loss from 60% to 50%. Therefore to receive the benefits of wildlife control, often the pest population must be reduced to an economic threshold level. To not reach this threshold may mean few benefits in terms of production, be a waste of resources, and there are also ethical issues surrounding the killing of wildlife with little benefit (O'Connor *et al.* 2005). To avoid this potential outcome in Tasmania it is important to implement a coherent package of management controls on native wildlife of concern regionally to both farming and plantation forestry (Coleman *et al.* 2006). For example, attempts to control population levels of native wildlife on farms and their impacts on pastures will fail unless this is done in coordination with related controls on animals using nearby public and private forestry plantations for shelter and as a source of food.

8.5 Future research

This study has been one of the most comprehensive studies of wildlife grazing on pastures in Tasmania. There is, however, the need for further research to address some of the important new questions to emerge during the study, and to consider some of the more significant limitations associated with the research. Some areas for future research may involve:

- *Determine if the Tasmanian pademelon is the main cause of reduced legume content in pastures*

It was hypothesised that the pademelon may have been the main wildlife species responsible for the observed reduction in pasture composition (cover abundance) of subterranean clover during pasture establishment experiments at Mt. Pleasant. This hypothesis is consistent with the diet preferences outlined by Dawson (1989). However, Statham (1992) found large proportions of clover in the guts of possums. Further research on this topic would require the use of experimental plantings of legume-rich pastures and the manipulation of grazing by pademelon and possums to evaluate this hypothesis.

- *Trialling the performance of pasture species that are tolerant of continued wildlife grazing in areas of high grazing pressure*

My study indicated that, of the four pasture types trialled, phalaris-based pastures appeared the most suited to establish and grow under heavy browsing pressure. There is opportunity to trial other pastures grasses, particularly grasses that are tolerant of continued grazing. New Hispanic cocksfoot and summer-active cultivars of cocksfoot have been planted for sheep grazing in the lower rainfall areas of Tasmania. An assessment of the capability of these new cultivars of pasture plant species and others to establish and grow under continued high wildlife grazing pressure is warranted.

- *Long term studies could be established to monitor the status of soil health under different grazing regimes*

It was apparent from the experiments conducted investigating the influence of wildlife grazing on soil health that wildlife either had no influence or that the duration and perhaps design of the experiment was inadequate. Longer term experiments may be required to adequately evaluate the influence of different grazing regimes on soil health.

- *Determine the potential competitive advantage of pasture plants located inside the exclusion cages used in this study*

An experiment could be undertaken to investigate if pasture plants growing within exclusion cages have a competitive advantage compared to pasture plant exposed to wildlife grazing. If a competitive advantage was conferred on pasture plants under the cages as a result of the experimental design then it is likely that the loss of pasture loss under heavy wildlife grazing may be over estimated. This experiment could be undertaken by using larger exclusion cages and larger buffers around cages.

- *Conduct studies in other areas of Tasmania*

Experiments could be conducted on Tasmania's east coast and south to quantify the potential impacts of wildlife grazing in those regions. Experiments conducted in the north-east and north-west of Tasmania by Norton *et al.* (2010) indicated that factors such as climate, landscape context and the wildlife species present in a region may influence the nature and extent of grazing impacts on pastures.

- *Quantify the impact of over abundant wildlife on native vegetation.*

This study has highlighted some of the effects of wildlife grazing on improved pastures. In fact, the improvement of pastures for agricultural livestock and food production has been implicated as one of the major factors leading to an over-abundance of native wildlife. There is an important need to address concerns that the over-abundance of some native wildlife species in several regions of Tasmania may be degrading native vegetation communities and impacting on biodiversity. Moreover, the increased installation of wallaby-proof fencing will increasingly restrict the movements of native wildlife and may result in serious animal welfare issues if food availability were to become critical and animals starve.

8.6 Conclusions

My study is the first to undertake a longer term evaluation of the influence of wildlife grazing on pastures in the Midlands region of Tasmania. Fosterville was considered representative of many pastoral properties in the region where significant areas of native vegetation have been retained. Hence, the observations reported in this study are likely to hold across (at least) many other properties of this region. Norton *et al.* (2010) reported similar results in relation to the impacts of wildlife grazing on pasture production at several farm sites in north-west and north-east Tasmania. These authors found that annual and seasonal pasture loss to wildlife grazing on farms could be high and the economic impost on farm profits could be significant. Pasture loss to wildlife grazing was inversely related to distance from the edge of native vegetation and this observed pattern was consistent with the relative abundance and habitat use by the wildlife examined (e.g. pademelon, wallaby, possum) (Norton *et al*. 2010).

It is clear that the anecdotal reports of farmers from the Midlands region and elsewhere in Tasmania are generally correct. Grazing by wildlife can result in significant pasture loss and may lead to a serious financial impost on farm enterprises. My study demonstrated the need to manage wildlife on farms and across regional landscapes in a holistic way in partnership with other land holders and land users. Given the variable nature of wildlife grazing in space and time and the controversial nature of wildlife control, this will be a complex and challenging task to implement.

References

ACT Government (2009) ACT Kangaroo Management Plan. ACT Government, Canberra, ACT.

Allen, P. G. (1987) Insect pests of pasture in perspective. In 'Temperate Pastures: their production, use and management.' (Eds JL Wheeler, CJ Pearson and GE Robards) pp. 211-225. (CSIRO, Melbourne, Australia)

Andrew, M. H. and Lange, R. T. (1986) The Spatial Distributions of Sympatric Populations of Kangaroos and Sheep - Examples of Dissociation Between These Species. *Wildlife Research* **13**(3), 367-373.

Archer, K. (1990) The effects of moisture supply and defoliation during flowering on seed production and hardseededness of *Trifolium subterraneum* L. *Australian Journal of Experimental Agriculture* **30**, 515-522.

Arias, M. E., Gonzalez-Perez, J. A., Gonzalez-Vila, F. J. and Ball, A. S. (2005) Soil health- a new challenge for microbiologists and chemists. *International Microbiology* **8**, 13-21.

Arnold, G. W., Steven, D. E. and Weeldenburg, J. R. (1989) The use of surrounding farmland by western grey kangaroos living in a remnant of wandoo woodland and their impact on crop production. *Australian Wildlife Research* **16**, 85-93.

Australian Government (2009) Lowland native grasslands of Tasmania. A nationally threatened ecological community. Policy Statement 3.18. Canberra, ACT, Environment Protection and Biodiversity Conservation Act 1999.

Australian Government (2010) Caring for Our Country Business Plan 2010-2011. Australian Government, Canberra, ACT.

Bailey, D. W., Gross, J. E., Laca, L. R., Rittenhouse, M. B., Coughenour, D. M., Swift, D. M. and Sims, P. L. (1996) Mechanisms that result in large herbivore grazing distribution patterns. *Journal of Range Management* **49**, 386-400.

Balota, E. L., Colozzi-Filho, A., Andrade, D. S. and Dick, R. P. (2003) Microbial biomass in soils under different tillage and crop rotation systems. *Biology and Fertility of Soils* **38**, 15-20.

Banks, P. B. (2001) Predation-sensitive grouping and habitat use by eastern grey kangaroos: a field experiment. *Animal Behaviour* **61**, 1013-1021.

Barlow, N. D. (1987) Pastures, pests and productivity: simple grazing models with two herbivores. *New Zealand Journal of Ecology* **10**, 43-55.

Barnes, A. and Hill, G. J. E. (1992) Estimating kangaroo damage to winter wheat crops in the Bungunya district of southern Queensland. *Wildlife Research* **19**(4), 417-427.

Bellotti, W. D. and Blair, G. J. (1985) Pasture establishment - Identifying the weak link. In 'Proceedings of the 3rd Australian Agronomy Conference. ' Ed. JJ Yates) pp. 178: University of Tasmania, Hobart, TAS)

Bentley, A. (1983) Fallow Deer. In 'The Australian Museum Complete Book of Australian Mammals.' Ed. R Strahan) pp. 501-502. (Angus and Robertson: Sydney)

Bestelmeyer, B. T., Herrick, J. E., Brown, J. R., Trujillo, D. A. and Havstad, K. M. (2004) Land management in the American Southwest: A State-and-Transition approach to ecosystem complexity. *Environmental Management* **34**, 38-51.

Bestelmeyer, B. T., Tugel, A. J., Peacock Jr., G. L., Robinett, D. G., Shaver, P. L., Brown, J. R., Herrick, J. E., Sanchez, H. and Havstad, K. M. (2009) State-and-Transition models for heterogeneous landscapes: A stategy for the development and application. *Rangeland Ecology and Management* **62**, 1-15.

Blair, G. (1997) Matching pastures to the Australian environment. In 'Pasture production and management.' (Eds JV Lovett and JM Scott) pp. 88-109. (Inkata Press: Melbourne)

Blanco, L. J., Ferrando, C. A. and Biurrun, F. N. (2009) Remote Sensing of Spatial and Temporal Vegetation Patterns in Two Grazing Systems. *Rangeland Ecology and Management* **62**(5), 445-451.

Blumstein, D. T. and Daniel, J. C. (2003) Foraging behaviour of three Tasmanian macropodid marsupials in response to present and historical predation threat. *Ecography* **26**, 585-594.

Bridle, K. L. and Kirkpatrick, J. B. (1999) Comparative effects of stock and wild vertebrate herbivore grazing on treeless subalpine vegetation, Eastern Central Plateau, Tasmania. *Australian Journal of Botany* **47**, 817-834.

Bridle, K. L. and Kirkpatrick, J. B. (2001) Impacts of grazing by vertebrate herbivores on the flower stem production of tall alpine herbs, Eastern Central Plateau, Tasmania. *Australian Journal of Botany* **49**, 459-470.

Briske, D., Derner, J., Brown, J., Fuhlendorf, S., Teague, R., Havstad, K., Gillen, R., Ash, A. and Willms, D. W. (2008) Rotational grazing on rangelands: reconciliation of perception and experimental evidence. *Rangeland Ecology and Management* **61**, 3-18.

Briske, D. D., Fuhlendorf, S. D. and Smeins, F. E. (2005) State-and-transition models, thresholds, and rangeland health: a synthesis of ecological concepts and perspectives. *Rangeland Ecology and Management* **58**, 1-10.

Brown, D. (1954) Weight estimate. In 'Methods of surveying and measuring vegetation.' pp. 97-125. (Commonwealth Agricultural Bureaux: Farnham Royal, Bucks, England)

Bulinski, J. and McArthur, C. (2000a) Observer error in counts of macropod scats. *Wildlife Research* **27**(3), 277-282.

Bulinski, J. and McArthur, C. (2000b) Spatial distribution of browsing damage and mammalian herbivores in Tasmanian eucalypt plantations. *Australian Forestry* **63**(1), 27-33.

Bünemann, E. K. and McNeill, A. Impact of fertilisers on soil biota. In 'Soil biology in agriculture: Proceedings of a workshop on current research into soil biology in agriculture', 2004, Tamworth Sustainable Farming Training Centre. (Ed. R Lines-Kelly), pp. 64-71

Calaby, J. H. (1983) Red-necked Wallaby. In 'The Australian Museum Complete Book of Australian Mammals.' Ed. R Strahan) pp. 239-241. (Angus and Robertson: Sydney)

Campbell, M. H., Hosking, W. J., Nicholas, D. A., Higgs, E. D. and Read, J. W. (1987) Establishment of perennial pastures. In 'Temperate pastures: their production, use and management.' (Eds JL Wheeler, CJ Pearson and GE Robards) pp. 59-74. (CSIRO Australia: Collingwood, VIC)

Carter, J. F. (1962) Herbage sampling for yield: tame pastures. In 'Pasture and range research techniques.' pp. 90-101. (Cornell University Press: Ithica, New York)

Caswell, H. (1978) Predator-Mediated Coexistence: A Nonequilibrium Model. *The American Naturalist* **112**(983), 127-154.

Caughley, G. (1964) Density and dispersion of two species of kangaroo in relation to habitat. *Australian Journal of Zoology* **12**(2), 238-249.

Caughley, G. (1977) 'Analysis of vertebrate populations.' (John Wiley & Sons Ltd: London, UK) 234 pp.

Caughley, G. (1982) Vegetation complexity and the dynamics of modelled grazing systems. *Oecologia* **54**, 309-312.

Caughley, G. (1987) Ecological relationships. In 'Kangaroos: their ecology and management in the sheep rangelands of Australia.' (Eds G Caughley, N Shepherd and J Short) pp. 159-187. (Cambridge University Press: Melbourne)

Caughley, G. and Lawnton, J. H. (1981) Plant-herbivore System. In 'Theoretical Ecology: Principles and Applications.' 2nd edn. Ed. RM May) pp. 132-166. (Blackwell Scientific: London)

Caughley, G., Shepherd, N. and Short, J. (1987) Kangaroos: their ecology and management in the sheep rangelands of Australia. In 'Kangaroos: their ecology and management in the sheep rangelands of Australia.'. pp. 252 pp.)

Caughley, J., Bomford, M., Parker, B., Sinclair, R., Griffiths, J. and Kelly, D. (1998) 'Managing Vertebrate Pests: Rodents.' (Bureau of Resource Sciences and Grains Research and Development Corporation, Canberra)

Chapman, D. F., Parsons, A. J., Cosgrove, G. P., Barker, D. J., Marotti, D. M., Venning, K. J., Rutter, S. M., Hill, J. and Thompson, A. N. (2007) Impacts of spatial patterns in pasture on animal grazing behavior, intake, and performance. *Crop Science* **47**, 399-415.

Choquenot, D., Lukins, B. and Curran, G. (1997) Assessing lamb predation by feral pigs in Australia's semi-arid rangelands. *Journal of Applied Ecology* **34**, 1445-1454.

Choquenot, D., McIlroy, J. and Korn, T. (1996) 'Managing Vertebrate Pests: Feral Pigs.' (Bureau of Resource Sciences, Australian Government Publishing Service, Canberra)

Claridge, A., Seebeck, J. and Rose, R. (2007) 'Bettongs, Potoroos and the Musky Rat-Kangaroo.' (CSIRO Publishing: Collingwood, VIC)

Clark, S. G. (1997) Subterranean Clover. In 'Technical Bulletin 47 - Grazing management of temperate pastures: literature reviews and grazing guidelines for major species.' (Eds RD Fitzgerald and GM Lodge) pp. 20-22. (NSW Agriculture: Orange, NSW)

Clarke, J. L., Welch, D. and Gordon, I. J. (1995) The influence of vegetation pattern on the grazing of heather moorland by red deer and sheep. 1. The location of animals on grass heather mosaics. *Journal of Applied Ecology* **32**, 166-176.

Close, D. J. and Davidson, N. (2004) Review of rural tree decline in a change Australian climate. *TasForests* **15**, 1-18.

Coleman, J. D., Montague, T. L., Eason, C. T. and Statham, H. L. (1997a) The management of problem browsing and grazing mammals in Tasmania. Department of Primary Industry and Fisheries, Tasmania.

Coleman, J. D., Montague, T. L., Eason, C. T. and Statham, H. L. (1997b) The management of problem browsing and grazing mammals in Tasmania. Revised by M Statham (2001). Tasmanian Institute of Agricultural Research, Launceston, Tasmania.

Coleman, J. D., Pech, R. P., Warburton, B. and Forsyth, D. M. (2006) Review of research into alternatives to the use of 1080 for management of browsing damage by mammals in Tasmania. Landcare Research New Zealand for the Department of Primary Industries and Water, Tasmania, No. LC/0506/144, Hobart.

Collins, W. J. (1978) Corrigendum - The effect of defoliation on inflorescence production, seed yield and hard-seededness in swards of subterranean clover. *Australian Journal of Agricultural Research* **29**(4), 789-801.

Commonwealth of Australia (1988) 'Zoological Catalogue of Australia - Mammalia.' (Australian Government Publishing Service: Bureau of Flora and Fauna, Canberra)

Community Leaders Group (2001) Tasmania Together Government activity report. Tasmania Together, Hobart, Tasmania.

Conlan, D. J., Dear, B. S. and Coombes, N. E. (1994) Effect of grazing intensity on herbage production and seed yields of *Trifolium subterraneum*, *Medicago murex*, and *Ornithopus compressus*. *Australian Journal of Experimental Agriculture* **34**, 181-188.

Connell, J. H. (1983) Interpreting the Results of Field Experiments: Effects of Indirect Interactions. *Oikos* **41**(2), 290-291.

Conquest, L. L. (2000) Analysis and Interpretation of Ecological Field Data Using BACI Designs: Discussion. *Journal of Agricultural, Biological, and Environmental Statistics* **5**(3), 293-296.

Cook, T. D. and Campbell D.T. (1979) 'Quasi-experimen-tation: design and analysis for field settings.' (Rand McNally: Chicago, Illinois, USA.)

Cotching, W. E. (2009a) Organic matter and soil life. In 'Soil health for farming in Tasmania.' pp. 35-45. (Bill Cotching: Devonport, Tasmania)

Cotching, W. E. (2009b) 'Soil Health for Farming in Tasmania.' (Bill Cotching: Devonport, Tasmania) pp.116

Coulson, G. M. and Raines, J. A. (1985) Methods for Small-Scale Surveys of Grey Kangaroo Populations. *Wildlife Research* **12**(2), 119-125.

Cowlishaw, S. J. (1955) The effect of sampling cages on the yields of herbage. *Grass and Forage Science* **6**(3), 179-182.

Craig, J. L., Mitchell, N. and Saunders, D. A. (2000) 'Nature Conservation 5. Conservation in Production Environments. Managing the Matrix.' (Surrey Beatty and Sons Pty Ltd.: New South Wales)

Croft, J. D., Fleming, P. J. S. and van de Ven, R. (2002) The impacts of rabbits on a grazing system in eastern New South Wales. 1. Ground cover in pastures. *Australian Journal of Experimental Agriculture* **42**, 909-916.

Crofts, F. C. (1997) Australian pasture production: The last 50 years. In 'Pasture production and management.' (Eds JV Lovett and JM Scott) pp. 1-16. (Inkata Press: Melbourne)

CSBP (2010) Short summary of all methods. CSBP Soil Analysis Laboratory, Perth.

Cullen, B. R., Eckard, R. J., Callow, M. N., Johnson, I. R., Chapman, D. F., Rawnsley, R. P., Garcia, S. C., White, T. and Snow, V. O. (2008) Simulating pasture growth rates in Australian and New Zealand grazing systems. *Australian Journal of Agricultural Research* **59**(8), 761-768.

Curtis, W. M. and Morris, D. I. (1994) 'The Student's Flora of Tasmania - Parts 1-4.' (St. David's Park Publishing: Hobart)

Daubenmire, R. F. (1940) Exclusure Technique in Ecology. *Ecology* **21**(4), 514-515.

Davis, N. E., Coulson, G. and Forsyth, D. M. (2008) Diets of native and introduced mammalian herbivores in shrub-encroached grassy woodland, south-eastern Australia. *Wildlife Research* **35**, 684-694.

Dawson, J. and Hocking, G. (2006) What's happening with our wildlife? In 'Tas-Regions. Vol. 12'. pp. 19-22)

Dawson, T. J. (1989) Diets of macropodoid marsupials: General patterns and environmental influences. In 'Kangaroos, Wallabies and Rat-Kangaroos. Vol. 1.' (Eds G Grigg, P Jarman and I Hume) pp. 129-142. (Surrey Beatty & Sons Pty Limited: Chipping Norton, NSW)

Decker, D. J. and Purdy, K. G. (1988) Toward a concept of wildlife acceptance capacity in wildlife management. *Wildlife Society Bulletin* **16**, 53-57.

Department of Primary Industries Parks Water and Environment (2010) Species of Game. In. ' (Department of Primary Industries, Parks, Water and Environment. http://www.dpiw.tas.gov.au/inter.nsf/WebPages/JCOK-68R49R?open: Hobart, Tasmania)

DeVries, W., Laca, A. E. and Demment, M. W. (1999) The importance of scale of patchiness for selectivity in grazing herbivores. *Oecologia* **121**, 355-363.

Dobb, J. L. and Elliott, C. R. (1964) Effect of pasture sampling cages on seed and herbage yields of creeping red fescue. *Canadian Journal of Plant Science* **44**, 96-99.

Dobbie, W. R., Berman, D. M. and Braysher, M. L. (1993) 'Managing Vertebrate Pests: Feral Horses ' (Bureau of Resource Sciences, Australian Government Publishing Service, Canberra)

Donaghy, D. J. and Fulkerson, W. J. (1998) Priority for allocation of water soluble carbohydrate reserves in *Lolium perenne* (L.). *Grass and Forage Science* **53**, 211-218.

Donaghy, D. J. and Tegg, R. (2001) Report of effects of wildlife on pasture growth at Elliot Research & Demonstration Station. Tasmanian Institute of Agricultural Research, Hobart.

Doran, J. W. and Safley, M. (1997) Defining and Assessing Soil Health and Sustainable Productivity. In 'Biological Indicators of Soil Health.' (Eds CE Pankhurst, BM Doube and VVSR Gupta) pp. 1-28. (CABI Publishing: Wallingford, UK)

Dorrough, J., Moxham, C., Crosthwaite, J. and Moll, J. (2005) *Vegetation Management in Temperate Livestock Production Landscapes*. Unpublished report to Land & Water Australia and Land Water & Wool. Department of Sustainability and Environment, Heidelberg.

Drewry, J. J. and Paton, R. J. (2005) Effects of sheep treading on soil physical properties and pasture yield of newly sown pastures. *New Zealand Journal of Agricultural Research* **48**, 39-46.

Driessen, M. M. and Hocking, G. J. (1992) Review and analysis of spotlight surveys in Tasmania: 1975-1990. Department of Parks, Wildlife and Heritage, No. 92/1, Hobart, Tasmania.

Dumont, B., Carrere, P. and D'Hour, P. (2002) Foraging in patchy grasslands: diet selection by sheep and cattle is affected by the abundance and spatial distribution of preferred species. *Animal Research* **51**(367-381).

Eberhardt, L. L. (1976) Quantitative ecology and impact assessment. *Journal of Environmental Management* **4**, 27- 70.

Eberhardt, L. L. (1978) Appraising variability in population studies. *Journal of Wildlife Management* **42**(2), 207-238.

Edwards, G. P., Croft, D. B. and Dawson, T. J. (1996) Competition between red kangaroos (*Macropus rufus*) and sheep (*Ovis aries*) in the arid rangelands of Australia. *Australian Journal of Ecology* **21**, 165-172.

Edwards, G. R., Newman, J. A., Parsons, A. J. and Krebs, J. R. (1994) Effects of the scale and spatial-distribution of the food resource and animal state on diet selection-an example with sheep. *Journal of Animal Ecology* **63**, 816-826.

Edwards, J. and Herridge, D. (2006) Soil-nitrate after drought. NSW Department of Primary Industries.

ERA (2010a) Broadacre Soil Health Test. The ERA Sustainable Farming Company, Malaga, WA.

ERA (2010b) ERA Bioassay Précis. The ERA Sustainable Farming Company, Malaga, WA.

Evans, M. C., Macgregor, C. and Jarman, P. J. (2006) Diet and feeding selectivity of common wombats. *Wildlife Research* **33**, 321-330.

Evans, P. S. (1973) The effect of repeated defoliation to three different levels on root growth of five pasture species. *New Zealand Journal of Agricultural Research* **16**, 31-34.

Fancy, S. G., Gross, J. E. and Carter, S. L. (2009) Monitoring the condition of natural resources in US national parks. *Environmental Monitoring and Assessment* **151**, 161-174.

Felsing, B., Glencross, B. and Telfer, T. (2005) Preliminary study on the effects of exclusion of wild fauna from aquaculture cages in a shallow marine environment. *Aquaculture* **243**(1-4), 159-174. [In English]

Fleming, P., Corbett, L., Harden, R. and Thomson, P. (2001) 'Managing the Impacts of Dingoes and Other Wild Dogs.' (Bureau of Rural Sciences, Canberra)

Fleming, P. J. S., Croft, J. D. and Nicol, H. I. (2002) The impact of rabbits on a grazing system in eastern New South Wales. 2. Sheep production. *Australian Journal of Experimental Agriculture* **42**, 917-923.

Fletcher, D. (2006) Population dynamics of eastern grey kangaroos in temperate grasslands. PhD Thesis, University of Canberra, Canberra

Foot, J. Z. and Romberg, B. (1965) The utilisation of roughage by sheep and the red kangaroo, *Macropus rufus* (Desmarest). *Australian Journal of Agricultural Research* **16**, 429-435.

Forsyth, D. M., Barker, R. J., Morriss, G. and Scroggie, M. P. (2007) Modeling the relationship between fecal pellets indices and deer density. *The Journal of Wildlife Management* **71**(3), 964-970.

Freemark, K. (1995) Assessing effects of agriculture on terrestrial wildlife: developing a hierarchical approach for the U.S. EPA. *Landscape and Urban Planning* **31**, 99-115.

Gatenby, S. (2010) Integrating financial and environmental data to improve farm business performance: an exploratory analysis of the Tasmanian beef industry. University of Tasmania, Hobart, Tas

Geisser, H. and Heinz-Ulrich, R. (2004) Efficacy of hunting, feeding, and fencing to reduce crop damage by wild boars. *Journal of Wildlife Management* **68**, 939-946.

Goh, C. S. S., Hodgson, D. R., Fearnside, S. M., Heller, J. and Malikides, N. (2005) Sodium monofluoroacetate (Compound 1080) poisoning in dogs. *Australian Veterinary Journal* **83**(8), 474-479.

Gong, W., Sinden, J., Braysher, M. and Jones, R. (2009) The economic impacts of vertebrate pests in Australia. Invasive Animals Cooperative Research Centre, Canberra.

Grant, C. (2006) State-and-Transition sucessional model for bauxite mining rehabilitation in Jarrah forest of Western Australia. *Restoration Ecology* **14**, 28-37.

Greentree, C., Saunders, G., McLeod, L. and Hone, J. (2000) Lamb predation and fox control in south-eastern Australia. *Journal of Applied Ecology* **37**(6), 935-943.

Greenwood, K. L. and Hutchinson, K. J. (1998) Root characteristics of temperate pasture in New South Wales after grazing at three stocking rates for 30 years. *Grass and Forage Science* **53**, 120-128.

Gregory, G. (1989) The control of pest wallaby populations - Masters Thesis. University of Tasmania, Hobart

Halsall, D. M., Leigh, J. H., Gollasch, S. E. and Holgate, M. D. (1995) The role of allelopathy in legume decline in pastures. II. Comparative effects of pasture, crop and weed residues on germination, nodulation and root growth *Australian Journal of Agricultural Research* **46**, 189-207.

Hegel, T. M., Gates, C. C. and Eslinger, D. (2009) The geography of conflict between elk and agricultural values in the Cypress Hills, Canada. *Journal of Environmental Management* **90**, 222-235.

Hendricks, H. H., Bond, W. J., Midgely, J. J. and Novellie, P. A. (2005) Plant species richness and composition a long livestock grazing intensity gradients in a Namaqualand (South Africa) protected area. *Plant Ecology* **176**, 19-33.

Hiernaux, P. (1998) Effects of grazing on plant species composition and spatial distribution in rangelands of the Sahel. *Plant Ecology* **138**, 191-202.

Hill, C. M. (1999) Conflict of Interest Between People and Baboons: Crop Raiding in Uganda. *International Journal of Primatology* **21**, 299-315.

Hill, G. J. E. (1978) Preliminary assessment of defacation patterns for eastern grey kangaroo. *Australian Zoology* **19**, 291-300.

Hill, G. J. E. (1981a) A Study of Grey Kangaroo Density Using Pellet Counts. *Wildlife Research* **8**(2), 237-243.

Hill, G. J. E. (1981b) A Study of Habitat Preferences in the Grey Kangaroo. *Wildlife Research* **8**(2), 245-254.

Hill, G. J. E., Barnes, A. and Wilson, G. R. (1988) The use of wheat crops by grey kangaroos, *Macropus giganteus* in Southern Queensland. *Australian Wildlife Research* **15**, 111-117.

Hill, J. O., Simpson, R. J., Moore, A. D., Graham, P. and Chapman, D. F. (2004) Impact of phosphorus application and sheep grazing on the botanical composition of sown pasture and naturalised, native grass pasture. *Australian Journal of Agricultural Research* **55**, 1213-1225.

Hoare, R. E. (1999) Determinants of human-elephant conflict in a land-use mosaic. *Journal of Applied Ecology* **36**, 689-700.

Holechek, J. L., Pieper, R. D. and Herbal, C. H. (1995) 'Range management. Principles and practices.' (Prentice Hall Englewood Cliffs, NJ, USA)

Holling, C. S. (1965) The functional response of predators to prey density and its role in mimicry and population regulation. *Memoirs of the Entomological Society of Canada* **45**, 1-62.

Hone, J. (2007) 'Wildlife damage control.' (CSIRO Publishing: Collingwood, Victoria) 179

How, R. A. (1983) Common Brushtail Possum. In 'The Australian Museum Complete Book of Australian Mammals.' Ed. R Strahan) pp. 147-148. (Angus and Robertson: Sydney)

Hurlbert, S. H. (1984) Pseudoreplication and the design of ecological field experiments. *Ecological Monographs* **54**(2), 187-2.

Hutchinson, K. J., King, K. L. and Wilkinson, D. R. (1995) Effects of rainfall, moisture stress, and stocking rate on the persistence of white clover over 30 years. *Australian Journal of Experimental Agriculture* **35**, 1039-1047.

IBM SPSS Inc. (2010) SPSS Statistics Release 19.0.0. In. ' Chicago)

Isbell, R. F. (2002) 'The Australian Soil Classification ' (CSIRO Publishing: Collingwood, Victoria) 144

Jackson, L. E. and Bloom, A. (1990) Root distribution in relation to soil nitrogen availability in field-grown tomatoes. *Plant and Soil* **128**(2), 115-126.

Jarman, P. J. and Phillips, C. M. (1989) Diets in a community of macropod species. In 'Kangaroos, Wallabies and Rat-Kangaroos. Vol. 1.' (Eds GC Grigg, P Jarman and ID Hume) pp. 143-149. (Surrey Beatty & Sons PTY Limited: Sydney)

Jeffery, S. J., Carter, J. O., Moodie, K. M. and Beswick, A. R. (2001) Using spatial interpolation to construct a comprehensive archive of Australian climate data. *Environmental Modelling and Software* **16/4**, 309-330.

Jenkinson, D. S. and Ladd, J. N. (1981) Microbial biomass in soil: measurement and turnover. In 'Soil Biochemistry. Vol. 5.' (Eds EA Paul and JN Ladd) pp. 415-471. (Marcel Dekker, Inc: New York and Basel)

Johnson, C. N. (1987) Macropod studies at Wallaby Creek. 4. Home range and movements of the red-necked wallaby. *Wildlife Research* **14**(2), 125-132.

Johnson, C. N., Jarman, P. J. and Southwell, C. J. (1987) Macropod studies at Wallaby Creek. 5. Patterns of defecation by eastern grey kangaroos and red-necked wallabies. *Wildlife Research* **14**(2), 133-138.

Johnson, I. R., Lodge, G. M. and White, R. E. (2003) The Sustainable Grazing Systems Pasture Model: description, philosophy and application to the SGS National Experiment. *Australian Journal of Experimental Agriculture* **43**, 711-728.

Johnson, K., Foster, S. and Foster, H. (2006) 'Fosterville', 'Merton Vale' and 'Bloomfield' Property Management Plan. Tasmanian Department of Primary Industries, Parks, Water and Environment, Hobart.

Johnson, K. A. and Rose, R. (1983) Tasmanian Pademelon. In 'The Australian Museum Complete Book of Australian Mammals.' Ed. R Strahan) pp. 226-227. (Angus and Robertson: Sydney)

Jones, R. M. (1992) Resting from grazing to reverse changes in sown pasture composition: application of the 'state-and-transition' model. *Tropical Grasslands* **26**, 97-99.

Jones, R. M., Jones, R. J. and McDonald, C. K. (1995) Some advantages of long-term grazing trials, with particular reference to changes in botanical composition. *Australian Journal of Experimental Agriculture* **35**, 1029-1038.

Jones, S., Lechner, A., Sheffield, K., Miura, N., Farmer, E., Reinke, K. and Norton, T. W. (2012) Integrating Science for Landscape Management. In 'Landscape Logic.' (Eds T Lefroy, A Curtis, A Jakeman and J McKee) pp. 189-202. (CSIRO Publishing: East Melbourne)

Kemp, D. R. (1999) Managing grassland composition with grazing. In 'Proceedings from the Bushcare Grassy Landscapes conference. ' (Eds T Barlow and R Thorburn) pp. 145-152. (Environment Australia: Clare, South Australia)

King, K. L. and Hutchinson, K. J. (1976) The effects of sheep stocking intensity on the abundance and distribution of mesofauna in pastures. *Journal of Applied Ecology* **13**(1), 41-55.

King, L. E., Lawrence, A., Douglas-Hamilton, I. and Vollrath, F. (2009) Beehive fence deters crop-raiding elephants. *African Journal of Ecology* **47**, 131-137.

Kioko, J., Muruthi, P., Omondi, P. and Chiyo, P. I. (2008) The performance of electric fences as elephant barriers in Amboseli, Kenya. *South African Journal of Wildlife Research* **38**, 52-58.

Kirkpatrick, J. B. and Bridle, K. L. (2007) 'People, sheep and nature conservation: the Tasmanian experience.' (CSIRO Publishing: Collingwood, Vic)

Kirkpatrick, J. B., Wildson, D., Meiss, A., Mollon, A. and Bridle, K. L. (2007) Chapter 4: Trees on the run. In 'People, sheep and nature conservation: the Tasmanian experience.' (Eds JB Kirkpatrick and KL Bridle) pp. 125-137. (CSIRO Publishing: Collingwood, VIC)

Knox, J. (1999) 'The glove-box guide to grass & legume identification in Tasmanian pastures.' (Department of Primary Industries, Water and Environment - Tasmania: Launceston)

Knox, J., Thompson and Campbell, S. (2006) 'Species for profit: A guide for Tasmanian pastures and field crops.' (Department of Primary Industries, Water and Environment - Tasmania: Launceston)

Krzic, M., Newman, R. F., Trethewey, C., Bulmer, C. E. and Chapman, B. K. (2006) Cattle grazing effects on plant species composition and soil compaction on rehabilitated forest landings in central interior British Columbia. *Journal of Soil and Water Conservation* **61**(3), 137-144.

Kücke, M., Schmid, H. and Spiess, A. (1995) A comparison of four methods for measuring roots of field crops in three contrasting soils. *Plant and Soil* **172**(1), 63-71.

Laca, E. A. (2008) Foraging in a heterogeneous environment: intake and diet choice. In 'Resource ecology: spatial and temporal dynamics of foraging.' (Eds HHT Prins and F van Langevelde) pp. 81-100. (Springer: Dordrecht, The Netherlands)

Laca, E. A. (2009) New Approaches and Tools for Grazing Management. *Rangeland Ecology and Management* **62**(5), 407-417.

Lacey, J. R., Jamtgaard, K., Riggle, L. and Hayes, T. (1993) Impacts of big game on private land in south-western Montana: Landowner perceptions. *Journal of Range Management* **46**(1), 31-37.

Landsberg, J., James, C. D., Morton, S. R., Müller, W. J. and Stol, J. (2003) Abundance and composition of plant species along grazing gradients in Australian rangelands. *Journal of Applied Ecology* **40**, 1008-1024.

Landsberg, J. and Wylie, F. R. (1988) Dieback of Rural Trees in Australia. *GeoJournal* **17**, 231-237.

le Mar, K. and McArthur, C. (2005) Comparison of habitat selection by two sympatric macropods, *Thylogale billardierii* and *Macropus rufogriseus rufogriseus*, in a patchy eucalypt-forestry environment. *Austral Ecology* **30**(6), 674-683.

Leigh, J. H., Halsall, D. M. and Holgate, M. D. (1995) The role of allelopathy in legume decline in pastures. I. Effects of pasture and crop residues on germination and survival of subterranean clover in the field and nursery. *Australian Journal of Agricultural Research* **46**, 179-188.

Leonard, S. (2009) Effects of Vertebrate Grazing and Environmental Factors on Fuel and Fire Potential in Tasmanian Native Grasslands. University of Tasmania, Hobart

Lesslie, R., Thackway, R. and Smith, J. (2010) *A National Level Vegetation Assets, States and Transitions (VAST) dataset for Australia (version 2)*. Bureau of Rural Sciences, Canberra.

Lodge, G. M. (1997) Phalaris. In 'Technical Bulletin 47 - Grazing management of temperate pastures: literature reviews and grazing guidelines for major species.' (Eds RD Fitzgerald and GM Lodge) pp. 8-9. (NSW Agriculture: Orange)

Lodge, G. M. and Murphy, S. R. (2006) Root depth of native and sown perennial grass based pastures, North-West Slopes, New South Wales. 1. Estimates from cores and effects of grazing treatments. *Australian Journal of Experimental Agriculture* **46**, 337-345.

Lodge, G. M. and Whalley, R. D. B. (1985) The manipulation of species composition of natural pastures by grazing management on the northern slopes of New South Wales. *Australian Rangeland Journal* **7**(1), 6-16.

Mapfumo, E., Naeth, M. A., Baron, V. S., Dick, A. C. and Chanasyk, D. S. (2002) Grazing impacts on litter and roots: perennial versus annual grasses. *Journal of Range Management* **55**(1), 16-22.

McDonald, D. and Rodges, D. (2010) Soils Alive! Understanding and Managing Soil Biology on Tasmanian Farms. Land Conservation Branch, DPIPWE, Hobart.

McDonald, R. C. and Isbell, R. F. (2009) Soil Profile. In 'Australian Soil and Land Survey: Field Handbook.' 3rd edn. pp. 147-204. (CSIRO Publishing: Melbourne)

McDonald, R. C., Isbell, R. F. and Speight, J. G. (2009) Land Surface. In 'Australian Soil and Land Survey: Field Handbook.' 3rd edn. pp. 127-145. (CSIRO Publishing: Melbourne)

McEachern, S., Francis, J., Lee, D. and Christie, J. (2009) Aginsights 2008. Holmes Sackett, Wagga Wagga, NSW.

McGregor, B. A. (2010) Influence of stocking rate and mixed grazing of Angora goats and Merino sheep on animal and pasture production in southern Australia. 1. Botanical composition, sward characteristics and availability of components of annual temperate pastures *Animal Production Science* **50**, 138-148.

McIntosh, D. L. (1966) The digestibility of two roughages and the rates of passage of their residues by the red kangaroo, *Megaleia rufa* (Desmarest), and the Merino sheep. *Wildlife Research* **11**, 125-35.

McIvor, J. G. and Gardener, C. J. (1995) Pasture management in the semi-arid tropical woodlands: effects on herbage yields and botanical composition. *Australian Journal of Experimental Agriculture* **35**, 705-715.

McLaren, C. (1997) Agriculture notes: Dry Sheep Equivalents for comparing different classes of livestock Department of Primary Industries, Victoria.

McNaughton, S. J. (1984) Grazing lawns: Animals in herds, plant form and coevolution. *The American Naturalist* **124**(6), 863-886.

McQuillan, P., Ireson, J., Hill, L. and Young, C. (2007) 'Tasmanian Pasture and Forage Pests: Identification, biology and control.' (Department of Primary Industries and Water, Tasmania)

McWilliam, J. R. (1968) The nature of the perennial response in Mediterranean grasses. II. Senescence, summer dormancy, and survival in *Phalaris*. *Australian Journal of Agricultural Research* **19**, 397-409.

Michaels, K., Norton, T., Lacey. M. and Williams, J. (2010) Spatial analysis of Tasmania's native vegetation cover and potential implications for biodiversity conservation. *Ecological Management and Restoration* **11**(3), 194-200.

Millennium Ecosystem Assessment (2005) Ecosystems and Human Well-Being: Synthesis. Island Press, Washington DC.

Milne, B. T., Turner, M. G., Wiens, A. J. and Johnson, A. R. (1992) Interactions between the fractal geometry of landscapes and allometric herbivory. *Theoretical Population Biology* **41**, 337-353.

Mitchell, N. and Craig, J. L. (2000) Managing the matrix: realigning paradigms toward sustainability. In 'Nature Conservation 5. Conservation in Production Environments. Managing the Matrix.' (Eds JL Craig, N Mitchell and DA Saunders) pp. 26-34. (Surrey Beatty and Sons Pty Ltd.: New South Wales)

Miura, N. and Jones, S. (2010) Characterising forest ecological structure using airborne laser scanner data. *Remote Sensing of Environment* **114**(5), 1069-1076.

Mokany, K., Friend, D., Kirkpatrick, J. and Gilfedder, L. (2006) 'Managing Tasmanian Native Pastures - a technical guide for graziers.' (Tasmanian Institute of Agricultural Research: Hobart)

Morgan, S. (1992) 'Land settlement in early Tasmania: Creating an antipodean England.' (CUP, Cambridge, UK)

Muhly, T. B. and Musiani, M. (2009) Livestock depredation by wolves and the ranching economy in the Northwestern U.S. *Ecological Economics* **68**, 2439-2450.

Munsell (1973) 'Munsell Soil Color Charts.' 1973 edn. (Munsell Color Co. Inc.: Baltimore, Maryland 21218, USA)

Murdoch, W. (1973) The functional response of predators. *Journal of Applied Ecology* **10**, 335-563.

Nagy, K. A., Sanson, G. D. and Jacobsen, N. K. (1990) Comparative field energetics of two macropod marsupials and a ruminant. *Australian Wildlife Research* **17**, 591-599.

Namoi CMA. (2010) *What is a Catchment Action Plan?* Namoi CMA, NSW. http://www.namoi.cma.nsw.gov.au/2010_cap_brochure_proof.pdf.

Neave, H. and Tanton, M. (1989) The Effects of Grazing by Kangaroos and Rabbits on the Vegetation and the Habitat of Other Fauna in the Tidbinbilla Nature Reserve, Australian Capital Territory. *Wildlife Research* **16**(3), 337-351.

Newsome, A. E. (1971) Competition between wildlife and domestic livestock. *Australian Veterinary Journal* **47**, 577-586.

Newsome, A. E. (1975) An ecological comparison of the two arid-zone kangaroos of Australia, and their anomalous prosperity since the introduction of ruminant stock to their environment. *The Quarterly Review of Biology* **50**(4), 389-424.

Nie, Z. N., Chapman, D. F., Tharmaraj, J. and Clements, R. (2004) Effects of pasture species mixture, management, and environment on the productivity and persistence of dairy pastures in south-west Victoria. 1. Herbage accumulation and seasonal growth pattern. *Australian Journal of Agricultural Research* **55**(6), 625-636.

Norton, T. W. (2010) Cradle Coast Regional Land Use Planning Initiative - Mapping Agricultural Land in the Cradle Coast region of Tasmania. Phase 2. Cradle Coast Authority, March, Burnie.

Norton, T. W. and Lacey, M. J. (2010) Patch Data Viewer - USER GUIDE. A decision support tool for the spatial analysis of native vegetation. Technical Report. CERF Landscape Logic National Research Hub, Hobart.

Norton, T. W., Lacey, M. J., Smith, R., Statham, M., Rawnsley, R., Donaghy, D., Gracie, A. and Burkitt, L. (2010) The nature and implications of browsing by native wildlife on Tasmanian farms. Tasmanian Institute of Agricultural Research, University of Tasmania, Hobart.

O'Connor, C., Warburton, B. and Fisher, M. Ethics and killing of wild sentient animals. In 'Proceedings of the 13th Australasian vertebrate pest conference', 2005, Wellington, New Zealand, pp. 203-207

O'Neill, R. V., Gardner, R. H., Milne, B. T., Turner, M. G. and Jackson, B. (1991) Heterogeneity and spatial hierarchies. In 'Ecological heterogeneity. Vol. 85-96.' (Eds J Kolasa and STA Pickett). (Springer-Verlag: New York, NY, USA)

Orr, D. M. and Evenson, C. J. (1991) Effects of Sheep Grazing Astrebla Grasslands in Central Western Queensland. Iii. Dynamics of Astrebla Spp. Under Grazing and Exclosure Between 1975 and 1986. *The Rangeland Journal* **13**(1), 36-46.

Owensby, C. E. (1969) Effect of Cages on Herbage Yield in True Prairie Vegetation. *Journal of Range Management* **22**(2), 131-132.

Oztas, T., Koc, A. and Comakli, B. (2003) Changes in vegetation and soil properties along a slope on overgrazed and eroded rangelands. *Journal of Arid Environments* **55**, 93-100.

Pajunen, A., Virtanen, R. and Roininen, H. (2008) The effects of reindeer grazing on the composition and species richness of vegetation in forest-tundra ecotone. *Polar Biology* **31**, 1233-1244.

Parker, V. T. and Pickett, S. T. A. (1997) Restoration as an ecosystem process: implications of the modern ecological paradigm. In 'Restoration Ecology and Sustainable Development.' (Eds KM Urbanska, NR Webb and PJ Edwards) pp. 17-32. (Cambridge University Press: Cambridge)

Parkes, J., Henzell, R. and Pickles, G. (1996) 'Managing Vertebrate Pests: Feral Goats.' (Australian Government Publishing Service, Canberra)

Parsons, A. J., Collett, B. and Lewis, J. (1984) Changes in the structure and physiology of a perennial ryegrass sward when released from a continuous stocking management: implications for the use of exclusion cages in continuously stocked swards. *Grass and Forage Science* **39**(1), 1-9.

Parsons, A. J. and Dumont, B. (2003) Spatial heterogeneity and grazing processes. *Animal Research* **52**, 161-179.

Parsons, A. J., Schwinning, S. and Carrere, P. (2001) Plant growth functions and possible spatial and temporal scaling errors in models of herbivory. *Grass and Forage Science* **56**, 21-34.

Pearson, C. J. and Jacobs, B. C. (1985) Root distribution in space and time in *Trifolium subterraneum*. *Australian Journal of Agricultural Research* **36**, 601-614.

Perry, R. J. and Braysher, M. L. (1986) A technique for estimating the numbers of eastern grey kangaroos, *Macropus giganteus*, grazing a given area of pasture. *Wildlife Research* **13**(3), 335-338.

Pickett, S. T. A., Collins, S. L. and Armesto, J. J. (1987) Models, mechanisms and pathways to sucession. *The Botanical Review* **53**, 335-371.

Pickett, S. T. A. and Parker, V. T. (1994) Avoiding the old pitfalls: opportunities in a new discipline. *Restoration Ecology* **2**, 75-79.

Poole, W. E. (1983) Eastern Grey Kangaroo. In 'The Australian Museum Complete Book of Australian Mammals.' Ed. R Strahan) pp. 244-247. (Angus and Robertson: Sydney)

Pople, A. R., Cairns, S. C., Menke, N. and Payne, N. (2006) Estimating the abundance of eastern grey kangaroos (*Macropus giganteus*) in south-eastern New South Wales, Australia. *Wildlife Research* **33**(2), 93-102.

Pratley, J. E. (1991) Pasture Management. In 'Australian sheep and wool handbook.' Ed. D Cottle) pp. 267-285. (Inkata Press: Melbourne)

Prober, S. M., Lunt, I. D. and Thiele, K. R. (2002) Determining reference conditions for management and restoration of temperate grassy woodlands: relationships among trees, topsoils and understorey flora in little-grazed remnants. *Australian Journal of Botany* **50**, 687-697.

Rao, K. S., Maikhuri, R. K., Nautiyal, S. and Saxena, K. G. (2002) Crop damage and livestock depredation by wildlife: a case study from Nanda Devi Biosphere Reserve, India. *Journal of Environmental Management* **66**, 317-327.

Rawnsley, R. P., Cullen, B. R., Turner, L. R., Donaghy, D. J., Freeman, M. and Christie, K. M. (2009) Potential of deficit irrigation to increase marginal irrigation response of perennial ryegrass (*Lolium perenne L.*) on Tasmanian dairy farms. *Crop & Pasture Science* **60**(12), 1156-1164.

Register of Australian Herbage Plant Cultivars (1972a) A. Grasses 1. Cockfoot *Dactylis glomerata L.* (Cocksfoot) cv. Porto. http://www.pi.csiro.au/ahpc/. Accessed 15-12-2012.

Register of Australian Herbage Plant Cultivars (1972b) A. Grasses 1. Cocksfoot *Dactylis glomerata L.* (Cocksfoot) cv. Currie. http://www.pi.csiro.au/ahpc/. Accessed 15-12-2012.

Register of Australian Herbage Plant Cultivars (1972c) A. Grasses 3. Phalaris *Phalaris aquatica L.* (Phalaris) cv. Australian. http://www.pi.csiro.au/ahpc/. Accessed 15-12-2012.

Reidy, M. M., Campbell, T. A. and Hewitt, D. G. (2008) Evaluation of electric fencing to inhibit feral pig movements. *Journal of Wildlife Management* **72**, 1012-1018.

Robertson, G. (1987) Plant dynamics. In 'Kangaroos: their ecology and management in sheep rangelands.' (Eds G Caughley, N Shepherd and J Short) pp. 50-68. (Cambridge University Press: Cambridge, UK)

Rossiter, R. C. (1961) The influence of defoliation on the components of seed yield in swards of subterranean clover (*Trifolium subterraneum* L.). *Australian Journal of Agricultural Research* **12**, 821-833.

Roughgarden, J. (1997) Production functions for ecological populations: a survey with emphasis on spatially explicit models. In 'Spacial Ecology.' (Eds D Tilman and P Kareiva) pp. 296-317. (Princeton University Press: Princeton, NJ)

Rumpff, L., Vesk, P., Duncan, D. and Wintle, B. (2010) *An Adaptive Approach to Vegetation and Biodiversity Management in the Goulburn Broken Catchment*. Landscape Logic Technical Report Number 12. Landscape Logic, Hobart.

Sanford, P., Cullen, B. R., Dowling, P. M., Chapman, D. F., Garden, D. L., Lodge, G. M., Andrew, M. Â H., Quigley, P. E., Murphy, S. R., King, W. M., Johnston, W. H. and Kemp, D. R. (2003) SGS Pasture Theme: effect of climate, soil factors and management onÂ pasture production and stability across the high rainfall zone of southern Australia. *Australian Journal of Experimental Agriculture* **43**(8), 945-959.

Sanson, G. D. (1978) The evolution and significance of mastication in the Macropodidae. *Australian Mammalogy* **2**, 23-28.

Sanson, G. D. (1989) Morphological adaptations of teeth to diets and feeding in the Macropodoidea. In 'Kangaroos, Wallabies and Rat-Kangaroos. Vol. 1.' (Eds G Grigg, P Jarman and I Hume) pp. 151-168. (Surrey Beatty & Sons Pty Limited: Chipping Norton, NSW)

SAS Institute Inc. (2004) SAS® 9.1.2 Qualification Tools User’s Guide. In. ' Cary, NC: SAS Institute Inc.)

Saul, G., Kearney, G. and Borg, D. (2009) Pasture systems to improve productivity of sheep in south-western Victoria. 1. Growth, composition, nutritive value and persistence of resown pastures *Animal Production Science* **49**, 654-667.

Saunders, G., Coman, B., Kinnear, J. and Braysher, M. (1995) 'Managing Vertebrate Pests: Foxes.' (Australian Government Publishing service, Canberra)

Schönbach, P., Wan, H., Gierus, M., Bai, Y., Müller, K., Lin, L., Susenbeth, A. and Taube, F. (2011) Grassland responses to grazing: effects of grazing intensity and management system in an Inner Mongolian steppe ecosystem. *Plant and Soil* **340**(1), 103-115.

Scott, J. J. and Kirkpatrick, J. B. (2008) Rabbits, landslips and vegetation change on the coastal slopes of subantarctic Macquarie Island, 1980-2007: implications for management. *Polar Biology* **31**(4), 409-419.

Scott, J. M. (1997) Pasture Establishment. In 'Pasture production and management.' (Eds JV Lovett and JM Scott) pp. 171-190. (Inkata Press: Melbourne)

Sherley, M. (2007) Is sodium fluoroacetate (1080) a humane poison? . *Animal Welfare* **16**, 449-458.

Short, J. (1985) The functional response of kangaroos, sheep and rabbits in an arid grazing system. *Journal of Applied Ecology* **22**, 435-447.

Short, J. (1986) The effect of pasture availability on food intake, species selection and grazing behaviour of kangaroos. *Journal of Applied Ecology* **23**, 559-571.

Short, J. (1987) Factors affecting food intake of rangelands herbivores. In ''Kangaroos: Their Ecology and Management in the Sheep Rangelands of Australia.' (Eds G Caughley, N Shepherd and J Short) pp. 84-98. (Cambridge University Press: Cambridge)

Smith, R. W., Statham, M., Norton, T. W., Rawnsley, R. P., Statham, H. L., Gracie, A. J. and Donaghy, D. J. (2012) Effects of wildlife grazing on the production, ground cover and plant species composition of an established perennial pasture in the Midlands region, Tasmania. *Wildlife Research* **39**(2), 123-136.

Southwell, C. (1989) Techniques for monitoring the abundance of kangaroo and wallaby populations. In 'Kangaroos, Wallabies and Rat-Kangaroos. Vol. 2.' (Eds GC Grigg, P Jarman and ID Hume) pp. 659-693. (Surrey Beatty and Sons PTY Limited: Chipping Norton, NSW)

Southwell, C. J. (1987) Macropod studies at Wallaby Creek. 2. Density and distribution of macropod species in relation to environmental variables. *Australian Wildlife Research* **14**(1), 15-33.

Sparling, G. P. (1997) Soil microbial biomass, activity and nutrient cycling as indicators of soil health. In 'Biological indicators of soil health.' (Eds CE Pankhurst, BM Doube and VVSR Gupta) pp. 97-120. (CABI Publishing: Wallingford, UK)

Spooner, P. G. and Allcock, K. G. (2006) Using a State-and-Transition approach to manage endangered *Eucalyptus albens* (White Box) woodlands. *Environmental Management* **38**, 771-783.

Sprent, J. A. and McArthur, C. (2002) Diet and diet selection of two species in the macropodid browser-grazer continuum: do they eat what they 'should'? *Australian Journal of Zoology* **50**(2), 183-192.

Statham, H., Statham, M. and Dawson, J. Alternatives to 1080 poison for control of native animals in Tasmania: A response to public concerns. In 'Proceedings from the 24th Vertebrate Pest Conference', 2010, Sacramento, CA. (Ed. RM Timm), pp. 97-103

Statham, H. L. (1992) Brushtail possums and rural dieback in Tasmania - A preliminary study. Midlands Tree Committee (Tas.), Oatlands, Tasmania.

Statham, M. (2000) Demonstration of the economic benefits to grazing from effective wallaby control, A report to the National Feral Animal Control Program, Bureau of Rural Sciences. Tasmanian Institute of Agricultural Research, Mt. Pleasant, Launceston.

Statham, M. (2005) The development of 1080 use for rabbit control in Tasmania. *Papers and Proceedings of the Royal Society of Tasmania* **139**, 1-6.

Statham, M. and Rayner, P. J. (1995) Loss of pasture and crop to native animals in Tasmania In '10th Australian Vertebrate Pest Conference. ' Ed. M Statham) pp. 171-176. (Department of Primary Industries and Fisheries, PO Box 46, Kings Meadows. Tasmania 7249: Hobart)

Statham, M. and Statham, H. L. (2009) 'Wallaby Proof Fencing : A planning guide for Tasmanian primary producers.' 1 edn. (Tasmanian Institute of Agricultural Research Hobart, Tasmania)

Statham, M. and Statham, H. L. (2010) 'Wallaby Proof Fencing: A planning guide for Tasmanian primary producers. Revised Edition.' (Tasmanian Institute of Agricultural Research: Hobart, Tasmania)

Steiner, J. J. and Grabe, D. F. (1986) Sheep Grazing Effects on Subterranean Clover Development and Seed Production in Western Oregon. *Crop Science* **26**, 367-372.

Stewart-Oaten, A. (2003) On rejection rates of paired intervention analysis: comment. *Ecology* **84**, 2795-2799.

Stewart-Oaten, A., Murdoch, W. W. and Parker, K. R. (1986) Environmental Impact Assessment: “pseudoreplication” in time? *Ecology* **67**, 929-940.

Swift, R. L. (1987) Large herbivore foraging and ecological hierarchies: landscape ecology can enhance traditional foraging theory. *BioScience* **37**, 789-799.

t' Mannetje, L. (1978) Measuring quantity of grassland vegetation. In 'Measurement of grassland vegetation and animal production.' Ed. L 't Mannetje) pp. 63-90. (Commonwealth Agricultural Bureaux, England: Farnham Royal, Bucks)

t' Mannetje, L. (2000a) 'Field and laboratory methods for grassland and animal production research.' (CABI: Wallingford) xii, 447 p.

t' Mannetje, L. (2000b) Measuring biomass of grassland vegetation. In 'Field and Laboratory Methods of Grassland and Animal Production Research.' (Eds L 't Mannetje and RM Jones) pp. 151-177. (CABI Publishing)

Tanner, Z. and Hocking, G. J. (2001) Status and management of the Forester Kangaroo in Tasmania 2000. Nature Conservation Report 2001/2002. Department of Primary Industries, Water and Environment.

Taylor, R. J. (1985) Effects of Pasture Improvement on the Nutrition of Eastern Grey Kangaroos and Wallaroos. *Journal of Applied Ecology* **22**(3), 717-725.

Terpstra, J. W. and Wilson, A. D. (1989) Grazing distribution of sheep and kangaroos in a semi-arid woodland. *Applied Animal Behaviour Science* **24**(4), 343-352.

Thackway, R. and Lesslie, R. (2006) Reporting vegetation condition using the Vegetation Assets, States and Transitions (VAST) approach. *Ecological Management and Restoration* **7**, S53-S62.

Triggs, B. (2009) 'Wombats.' 2nd edn. (CSIRO Publishing: Collingwood, Vic)

van den Berg, H. and Cock, M. J. W. (1993) Exclusion cage studies on the impact of predation on Helicoverpa armigera in cotton. *Biocontrol Science and Technology* **3**(4), 491-497.

Van Zwieten, L. (2004) Impacts of pesticides on soil biota. In 'Soil biology in agriculture: Proceedings of a workshop on current research into soil biology in agriculture. ' Ed. R Lines-Kelly) pp. 72-79. (NSW Department of Primary Industries: Tamworth Sustainable Farming Training Centre)

Veblen, K. E. and Young, T. P. (2010) Contrasting effects of cattle and wildlife on the vegetation development of a savanna landscape mosaic. *Journal of Ecology* **98**, 993-1001.

Victorian Government Department of Primary Industries (2005) Sustainable Carrying Capacity: Monitoring tools. Victorian Government Department of Primary Industries.

Virgona, J. M. and Hill, A. J. (1997) Cocksfoot. In 'Technical Bulletin 47 - Grazing management of temperate pastures: literature reviews and grazing guidelines for major species.' (Eds RD Fitzgerald and GM Lodge) pp. 12-13. (NSW Agriculture: Orange)

Volaire, F. (1994) Effects of summer drought and spring defoliation on carbohydrate reserves, persistence and recovery of two populations of cocksfoot (*Dactylis glomerata*) in a Mediterranean environment. *Journal of Agricultural Science, Cambridge* **122**, 207-215.

Wahungu, G. M., Catterall, C. P. and Olsen, M. F. (2001) Predator avoidance, feeding and habitat use in the red-necked pademelon, *Thylogale thetis*, at rainforest edges. *Australian Journal of Zoology* **49**(1), 45-58.

Walker, J. W. and Heitschmidt, R. K. (1986) Effects of various grazing treatments on type and density of cattle trails. *Journal of Range Management* **39**(428-431).

Wardell-Johnson, G. and Williams, M. (2000) Edges and gaps in mature karri forest, south-western Australia: logging effects on bird species abundance and diversity. *Forest Ecology and Management* **131**, 1-21.

Wardlaw, T. and Burton, D. (2008) Targeted culling of browser species that most threaten eucalypt plantations. Forestry Tasmania, Hobart.

Weber, G. E., Jeltsch, F., Van Rooyen, N. and Milton, S. J. (1998) Simulated long-term vegetation response to grazing heterogeneity in semi-arid rangelands. *Journal of Applied Ecology* **35**, 687-699.

Westoby, M., Walker, B. H. and Noy-Meir, I. (1989) Opportunistic management for rangelands not at equilibrium. *Journal of Range Management* **42**, 266-274.

Whalley, R. D. B. and Bellotti, B. (1997) Natural pastures and rangelands. In 'Pasture production and management.' (Eds JV Lovett and JM Scott) pp. 110-132. (Inkata Press: Melbourne)

Whalley, R. D. B. and Lodge, G. M. (1987) Use of native and natural pastures. In 'Temperate pastures: their production, use and management.' (Eds JL Wheeler, CJ Pearson and GE Robards) pp. 533-550. (Australian Wool Corporation/CSIRO)

Wheeler, J. L. (1987) Pastures and pasture research in southern Australia. In 'Temperate Pastures: their production, use and management.' (Eds JL Wheeler, CJ Pearson and GE Robards) pp. 3-31. (Australian Wool Corporation/CSIRO)

While, G. M. and McArthur, C. (2005) Foraging in a risky environment: a comparison of Bennett's wallabies *Macropus rufogriseus rufogriseus* (Marsupialia:

Macropodidae) and red-bellied pademelons *Thylogale billiardierii* (Marsupialia: Macropodidae) in open habitats. *Austral Ecology* **30**(7), 756-764.

While, G. M. and McArthur, C. (2006) Distance from cover affects artificial food-patch depletion by macropod herbivores. *Wildlife Research* **33**, 565-570.

Williams, K., Parer, I., Coman, B., Burley, J. and Braysher, M. (1995) 'Managing Vertebrate Pests: Rabbits.' (Bureau of Resource Sciences, Australian Government Publishing Service, Canberra)

Williams, S. S. (1951) Microenvironment in relation to experimental techniques. *Grass and Forage Science* **6**(4), 207-217.

Wissmar, R. C. and Bisson, P. A. (2003) Strategies for restoring rivers: problems and opportunities. In 'Strategies for restoring river ecosystems: sources of variability and uncertainty in natural and management systems.'. (Eds RC Wissmar and PA Bisson) pp. 245-262: American Fisheries Society, Bethesda, Maryland.)

Yates, C. J. and Hobbs, R. J. (1997) Woodland restoration in the Western Australian wheatbelt: a conceptual framework using a state and transition model. *Restoration Ecology* **5**, 28-35.

Yates, C. J., Norton, D. A. and Hobbs, R. J. (2000) Grazing effects on plant cover, soil and microclimate in fragmented woodlands in south-western Australia: implications for restoration. *Austral Ecology* **25**, 36-47.

Zar, J. H. (1999) 'Biostatistical Analysis. Fourth edition.' (Upper. Saddle River, NJ: Prentice Hall) 663 pp.

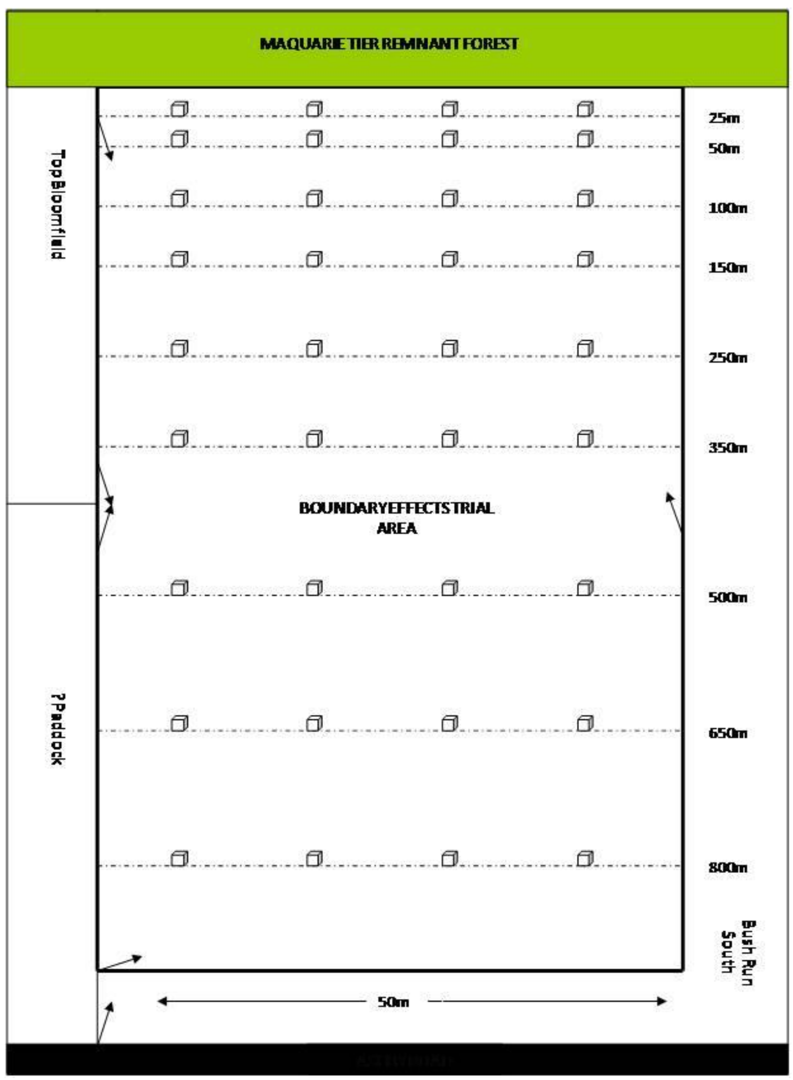
Zedler, J. (2005) Ecological restoration: guidance from theory. *San Francisco Estuary and Watershed Science* **3**(2), 1-31.

Zhou, H., Tang, Y., Zhao, X. and Zhou, L. (2006) Long-term grazing alters species composition and biomass of a shrub meadow on the Qinghai-Tibet Plateau. *Pakistan Journal of Botany* **38**(4), 1055-1069.

Appendices

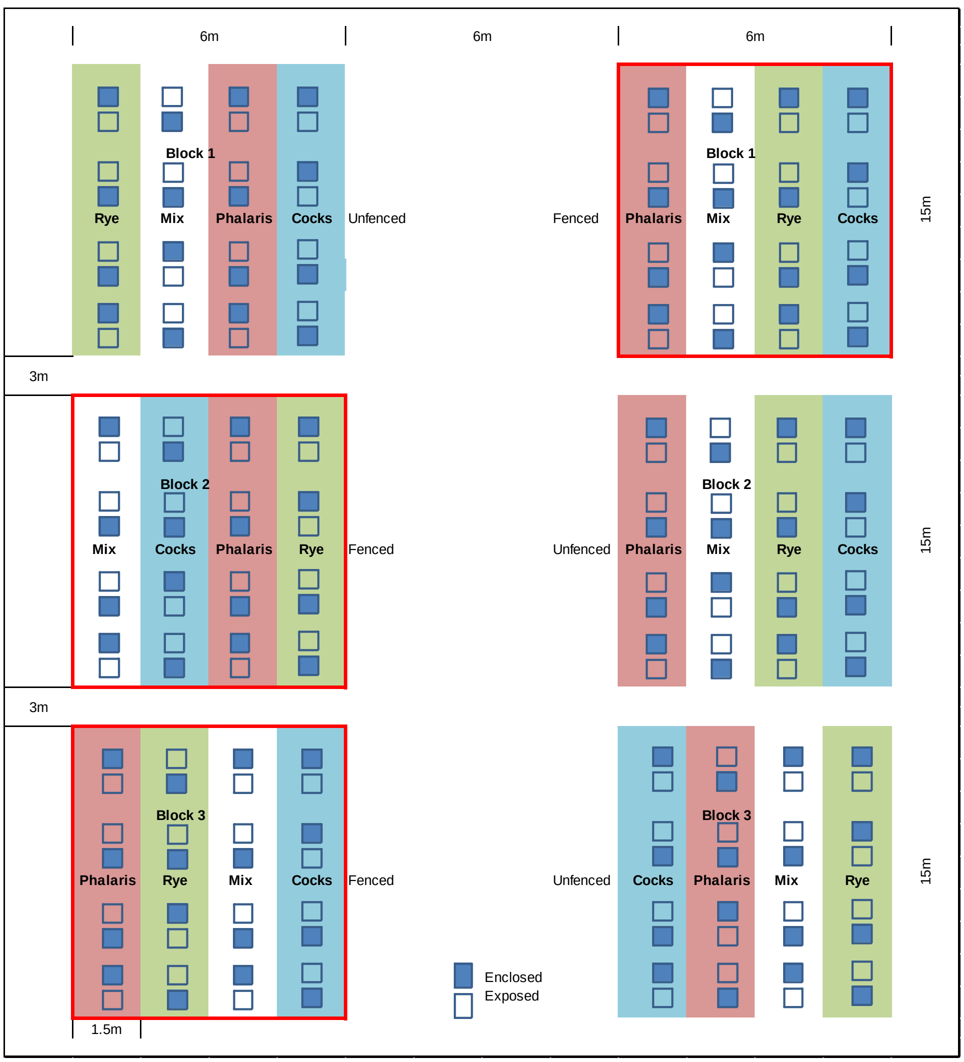
Appendix 1- Site plan of experiments undertaken in Chapters 3, 4 and 5

The trial design below refers to the experiments undertaken in Chapter 3, 4 and 5 at Fosterville. The experimental site was 900 m in length and 50 m in width. Four exclusion cages (represented by boxes) were erected at each of the nine distance boundaries marked on the right hand side of the figure. A paired exposed to wildlife grazing plot was marked next to each of the exclusion cages for comparison of measured variables. The green shaded area represents the native vegetation on the Macquarie Tier, as depicted in Figure 3.1.



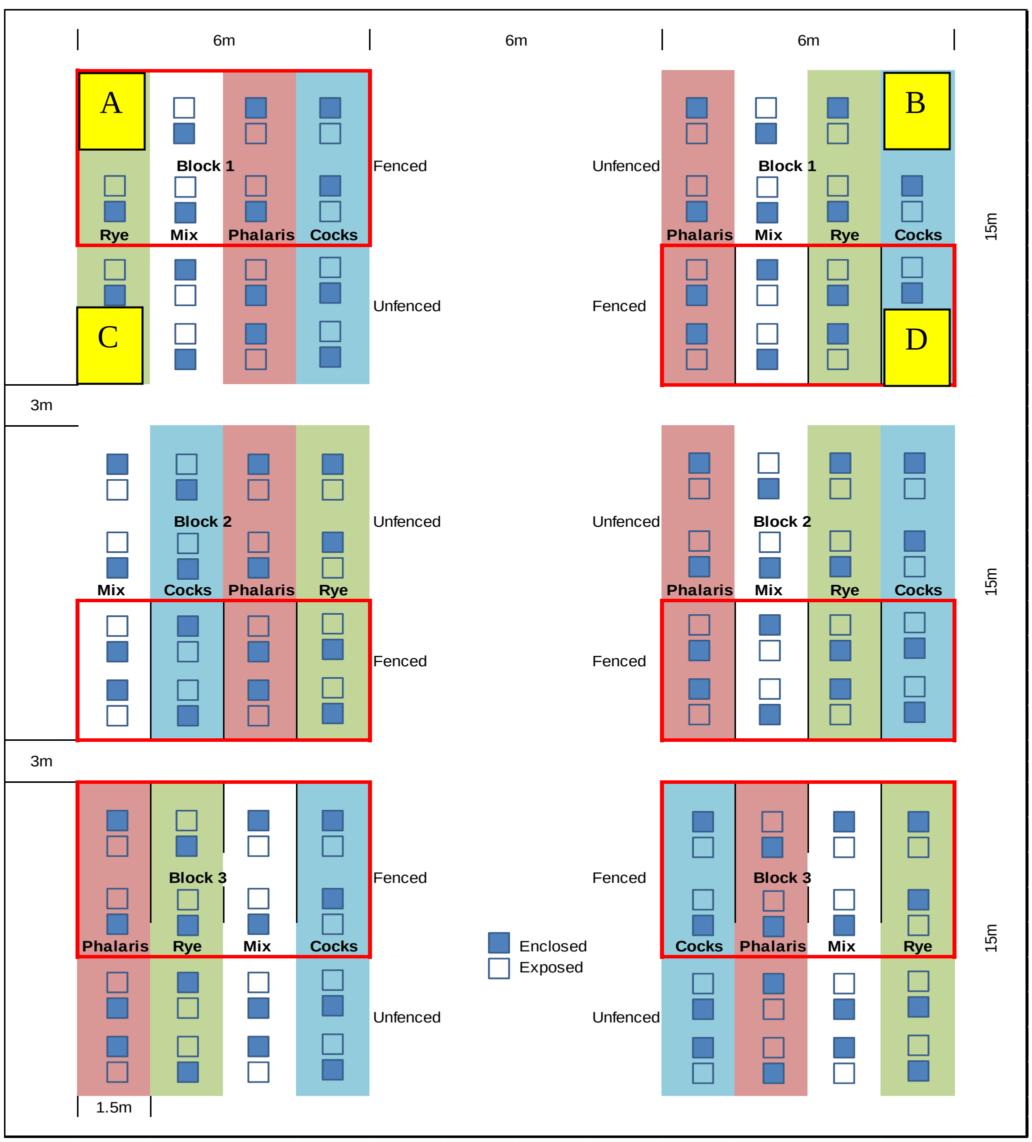
Appendix 2 - Site plan of experiments undertaken in Chapter 6

The trial design below refers to the experiments undertaken in Chapter 6 at Fosterville, with a similar design implemented at Mt. Pleasant. There are three blocks, each with one fenced (red outlined) and unfenced main treatment. Within each fencing treatment, four pasture type (coloured columns) sub plot treatments were sown. Within each pasture type treatment, four exclusion cages (solid boxes) were erected with four marked uncaged areas (empty boxes). The uncaged plots were used to assess the impact of grazing on pasture growth and production, while the caged plots were used to assess the impact of the cages on pasture growth. This is also the design of the 2009 fencing treatment outlined in Chapter 7.



Appendix 3 - Site plan of experiments undertaken in Chapter 7

The trial design below refers to the experiments undertaken in Chapter 7. This experiment follows on from the one undertaken in Chapter 6 (see appendix 2). In this experiment the fencing treatments were modified so that there were two fenced and two unfenced areas within each block, outlined in red (below). This meant that within each block there were four fencing treatments: a) unfenced 2009/fenced 2010; b) fenced 2009/unfenced 2010; c) unfenced 2009/unfenced 2010; and d). fenced 2009/ fenced 2010.



Appendix 4 - Wildlife Research paper

The results of Chapters 3 and 4 have been published in a peer-reviewed international journal (see citation below) and the paper is attached in this appendix. The text may vary slightly between the thesis and the paper. In addition, a manuscript of Chapters 5 and 6 is being finalised.

Citation:

Smith RW, Statham M, Norton TW, Rawnsley RP, Statham HL, Gracie AJ, Donaghy DJ (2012) Effects of wildlife grazing on the production, ground cover and plant species composition of an established perennial pasture in the Midlands region, Tasmania. *Wildlife Research* **39**(2), 123-136.